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LAND APPLICATION OF LIQUID WASTE:
A CASE STUDY OF PIGGERY WASTE APPLICATION
ON LIGHT SOILS

A thesis
submitted in partial fulfilment
of the requirement for the degree
of
Master of Applied Science
in
Natural Resources Engineering

at the
University of Canterbury

by

S P Pandey

Lincoln University
1990
ABSTRACT

This study was conducted on soil from the Burnham piggery which is located on the Alluvial plains of mid-Canterbury between the Waimakariri and Selwyn rivers. Effluent from the piggery is spray-irrigated onto the adjacent land. Soils at the disposal sites are flat, somewhat excessively drained. Subsurface drainage of effluent has a measurable impact on the quality of groundwater.

The case study was designed to determine the effects of land application of piggery waste on the contamination of groundwater. The amounts of NO$_3^-$-N and NH$_4^+$-N leached from small undisturbed soil lysimeters (180 mm dia * 150-200 mm deep) were compared under a standard solution application and a slurry application.

Examination of breakthrough curves from solution experiments indicated extensive preferential flow of solute had occurred through natural soil macropores.

The initial peak recovery of N in the standard solution experiment appeared quickly (after 0.3 pore volume) and about 80% of the N applied was recovered by 1.0 pore volume of drainage. The rate of recovery then decreased and became constant. Total recovery of N in the leachate equalled approximately 100% following the application of an amount of water equivalent to the amount of rainfall that would occur during an average winter period.

The total mean percent recovery of N in the slurry experiment was fairly evenly distributed and about 35% of the total N in the leachate was recovered by 1.0 pore volume of drainage. The total recovery of N in the leachate equalled about 8% following the application of an amount of water equivalent to the amount of rainfall during an average winter period.

About 76% of applied N was recovered from the soil. Results were also predicted assuming the soil to be at field capacity and permanent wilting point at the time of slurry application and the start of leaching. Cases of maximum rainfall and evapotranspiration, and, minimum rainfall and evapotranspiration over the period were also considered.
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CHAPTER 1
INTRODUCTION

It is common in New Zealand to discharge waste materials, either raw or treated, into natural waters; coastal and inland waters. This is putting stress on such habitats and is also offending the cultural and spiritual values of many people.

There is increasing interest in the use of land for the management of various types of waste products, both solid and liquid. It is being realised that considerable cost can be saved and the quality of the environment enhanced by using the land in an appropriate manner for waste treatment and/or utilisation. Land application is an option for final disposal/utilisation of a waste that can enhance rather than diminish the environment if such systems are designed and managed correctly.

The types of waste product may be raw, treated, solid, semi-solid, semi-liquid or liquid. They vary from industrial wastes, municipal wastes to farm animal wastes.

There are a variety of methods used to apply wastes, ranging from subsurface injection, flooding, sprinkler irrigation and solid spreading.

Land application of liquid waste from piggeries and dairy farms is being increasingly used on pastoral farms in New Zealand. After applying piggery waste that contains a high proportion of nitrogen in the form of $\text{NH}_4^+$-N and urea, however, the process of nitrification can convert $\text{NH}_4^+$-N into $\text{NO}_3^-$-N which is very mobile and susceptible to leaching losses.

Leaching losses of nitrate (following the wastewater application) can cause: (a) adverse environmental effects including toxic nitrate levels in drinking water and eutrophication of lakes and rivers and (b) large amount of nitrogen being leached out of the soil rooting zone resulting in the reduced plant production and inefficient use of nitrogen (Ball, 1979).

Leaching losses of $\text{NO}_3^-$ to groundwater occur principally when soil $\text{NO}_3^-$ levels are high and downward water movement is large. The magnitude of such losses depends on factors such as rainfall, evaporation, soil type and plant cover.

The main objectives of the work presented in this thesis were:

1. To complete a detailed review of land application of waste material, covering the methods used, health and environmental hazards, social and cultural impacts, and soil, soil water and waste interactions.
2. To conduct a case study on a large piggery unit near Burnham, sited on stony, free draining soil over a sensitive groundwater source which would quantify the potential loss of N and the effect on groundwater due to land disposal of pig effluent.
CHAPTER 2
LITERATURE REVIEW
LAND APPLICATION OF WASTES

2.1 Introduction

The land represents not only an appropriate disposal medium for wastes but also an opportunity to manage the wastes with minimal adverse environmental effects (Loehr et al., 1976). Critical evaluation of those factors that impose specific limitations on the land application of wastes is needed. Specifically, it is necessary to evaluate:

a: the characteristics of the waste water
b: the impact of nitrogen, phosphorus and potentially toxic elements on the soil system, vegetative cover, runoff water and ground water
c: the interactions between the soil system and waste water constituents
d: the social and legal constraints
e: the effect of the proposed land application system on present and future regional land-use patterns (Loehr et al., 1979a)

Waste products from industry, townships and primary production will always be with us. The challenge is to manage them in a way that conserves rather than destroys our natural environment. Land application can be one of those methods if done correctly.

2.2 Methods of Land Application

Land application/treatment is defined as the controlled application of the waste water onto the land surface to achieve a designed degree of treatment through physical, chemical and biological process within the plant-soil-water system.

There are four main objectives in applying liquid waste onto the land, and in considering any proposal the principal intentions should be clearly established. It may be:

a: a means of treatment for improving effluent quality;
b: a means of disposal of effluents;
c: a form of irrigation to supply moisture for plant growth;
There are several land application processes by which these objectives can be achieved (Figure 2.1).

2.2.1 Slow Rate Process or Irrigation Treatment

Slow rate process is the application of the waste water to a vegetated land surface with the applied waste water being treated as it flows through the plant-soil matrix. The flow is slow and infiltration and percolation occur within the application site. Surface application techniques include ridge-and-furrow and border strip flooding. Application by sprinklers can be from fixed risers or from moving systems. This process is of particular importance where the purified water may finally enter waterways or important aquifer systems. When requirements are very stringent for nitrogen, phosphorus, biochemical oxygen demand (BOD), suspended solids (SS), pathogens, metals and trace organics, they can be met usually with a slow rate process. This system is capable of producing the highest degree of waste water treatment of all land treatment systems.

2.2.2 Rapid Infiltration Process

In rapid infiltration land treatment most of the applied waste water percolates through the soil, and the treated effluent drains naturally to surface waters or joins the ground water. The waste water is applied to moderately and highly permeable soils (such as sands and loamy sands), by spreading in basins or by sprinkling, and is treated as it travels through the soil matrix. Purification may be minimal and the system is usable only where the quality of the receiving water is not critical.

2.2.3 Overland Flow Process

In overland flow treatment, waste water is applied at the upper reaches of grass covered slopes and allowed to flow over the vegetated surface to runoff collection ditches. In this process only a small amount of infiltration occurs. This process may achieve only limited nutrient removal but the organic and suspended solids content of the waste water is reduced.

Typical design features for the three land application processes are compared in Table 2.1. The major soil characteristics are compared for each process in Table 2.2 (these are seen as desirable characteristics and not limits to be adhered to rigorously).
Figure 2.1 Land application processes: (a) Irrigation treatment; (b) Overland flow treatment; (c) Rapid infiltration. (From Stevenson, 1976)
2.2.4 Other Methods of Land Application of Liquid Waste

2.2.4.1 Subsurface Injection of Liquid Waste

Subsurface injection of liquid waste is the incorporation of waste, either agricultural or municipal, into the aerobic layer of soil. Subsurface injection of wastewater to non-cultivated land is practiced only to a limited extent. The main reasons are the generally rougher terrain and the presence of a permanent vegetation (e.g. shrubs, trees and forests). The areas most suitable for subsurface injection are therefore limited. Such areas as grazing land, bare lake beds and playgrounds offer some possibilities. Subsurface injection has the following advantages:

a. immediate covered disposal of waste;
b. elimination of odour, flies and vermin;
c. placement in rootzone for plants;
d. control of surface runoff and loss by soil and water erosion; and
e. complete containment of pathogenic micro-organism can readily hold the pollution to a minimum.

However the method has several disadvantages:

a. the need for complex management;
b. requirement for special equipment with concomitant cost; and
c. application problems in wet soils.

Warner et al. (1987) have designed injector tines so that the volume of voids created by tines is sufficient to accommodate the application rate. Godwin and Spoar (1977) designed simple narrow tines which produced very localised soil disturbances, hence the volume of voids readily available was essentially that displaced by the tine itself. Later on Negi et al. (1978) and Spoar and Godwin (1978) showed that the addition of wings to the tines produced a greater increase in the volume of the soil disturbance than the corresponding increase in the draught force. The winged tine also distributed the sludge more uniformly across the soil and sufficient soil cover was created to prevent both surface pollution and unpleasant smell. Choudhary et al. (1988) designed a liquid waste injector machine for shallow sub-soil placement. The injector incorporates the opener concept (inverted T-opener, originally designed for direct drilling) in order to reduce the loss of nitrogen through volatilisation of the material being injected.
Table 2.1 Comparison of Typical Design Features for Land Treatment Processes
(adapted from EPA, 1981)

<table>
<thead>
<tr>
<th>Feature</th>
<th>Slow Rate</th>
<th>Rapid Infiltration</th>
<th>Overland Flow</th>
</tr>
</thead>
<tbody>
<tr>
<td>Application techniques</td>
<td>Sprinkler or surface&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Usually surface</td>
<td>Sprinkler or surface</td>
</tr>
<tr>
<td>Annual loading rate, m</td>
<td>0.5-6</td>
<td>6-125</td>
<td>3-20</td>
</tr>
<tr>
<td>Field area required, ha&lt;sup&gt;b&lt;/sup&gt;</td>
<td>23-280</td>
<td>3-23</td>
<td>6.5-44</td>
</tr>
<tr>
<td>Typical weekly loading rate, cm</td>
<td>1.3-10</td>
<td>10-240</td>
<td>6-40&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>Minimum pre-application treatment provided in the USA</td>
<td>Primary sedimentation&lt;sup&gt;d&lt;/sup&gt;</td>
<td>Primary Sedimentation&lt;sup&gt;e&lt;/sup&gt;</td>
<td>Grit removal Comminution&lt;sup&gt;e&lt;/sup&gt;</td>
</tr>
<tr>
<td>Disposition of applied wastewater</td>
<td>Evapotranspiration &amp; percolation</td>
<td>Mainly percolation</td>
<td>Surface runoff and evapotranspiration with some percolation</td>
</tr>
<tr>
<td>Need for vegetation</td>
<td>Required</td>
<td>Optional</td>
<td>Required</td>
</tr>
</tbody>
</table>

- Includes ridge-and-furrow and border strip.
- Field area in hectare not including buffer area, roads, or ditches for 3,785 m³/d.
- Range includes raw wastewater to secondary effluent, higher rates for higher level of pre-application treatment.
- With restricted public access; crops not for direct human consumption.
- With restricted public access.
### Table 2.2 Comparison of Site Characteristics for Land Treatment Processes
(adapted from EPA, 1981)

<table>
<thead>
<tr>
<th></th>
<th>Slow rate</th>
<th>Rapid infiltration</th>
<th>Overland flow</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Grade</strong></td>
<td>&lt;20% on cultivated land; &lt;40% on noncultivated land</td>
<td>Not critical; excessive grades required much earthwork</td>
<td>Finish slopes 2-8%a</td>
</tr>
<tr>
<td><strong>Soil permeability</strong></td>
<td>Moderately slow to moderately rapid</td>
<td>Rapid (sands, sandy loams)</td>
<td>Slow (clay, silt and soil with impermeable barriers)</td>
</tr>
<tr>
<td><strong>Depth to groundwater</strong></td>
<td>0.6-1 m (minimum)b</td>
<td>1 m during flood cycleb; 1.5-3 m during drying cycle</td>
<td>Not criticalc</td>
</tr>
<tr>
<td><strong>Climatic restriction</strong></td>
<td>Storage often needed for cold weather and during heavy precipitation</td>
<td>None (possibly modify operation in cold weather)</td>
<td>Storage usually needed for cold weather</td>
</tr>
</tbody>
</table>

---

**Notes:**

a. Steeper grades might be feasible at reduced hydraulic loading.
b. Underdrains can be used to maintain this level at sites with high groundwater table.
c. Impact on groundwater should be considered for more permeable soils.

---

### 2.2.4.2 Sand-Filters

Sand-filters are often used for the decomposition of organic pollutants such as liquid animal waste. For this it is necessary that aerobic conditions prevail, hence sandy soil is often preferred for filters. A recommendation of suitable sand particle size in the range of 0.2-0.5 mm was cited by Kowalik and Obarska-Pempkowiak (1985). Liquid waste such as the supernatant from pig manure slurry (after removal of suspension) are usually piped to the irrigation site and flood-irrigated into the basins. Field drains embedded in gravel are installed in the subsoil to collect surplus drainage (Figure 2.2). Investigation by Kristiansen (1981) showed that after loading two trenches with septic tank effluent, the effluent concentrated in one end of the filters, resulting in the clogging of the sand surface and subsequent ponding. Increased clogging intensity induced a lower soil water content below the crust-zone. Because the filters were aerated from below, the increased clogging led to improved aeration.
2.3 The Nitrogen Cycle

The Nitrogen Cycle constitutes the gain and the loss of N within the soil/plant system, accompanied by the complex biochemical transformations.

In some areas, septic tanks are suspected to cause an increase in the content of groundwater nitrogen. For that reason, Kristiansen (1961) investigated the turnover of N in sand filter trenches. The total N content in the soil ranges from more than 2.5% in peats to less than 0.02% in subsoils (Bremner, 1965). The N thus found in soils can be categorized into three major forms: (a) organic-N associated with soil humus; (b) ammonium-N fixed by some clay minerals; (c) soluble inorganic N compounds often called Mineral-N (i.e., NH₄⁺, NO₂⁻, and NO₃⁻). Mineral-N represents the only form of nitrogen which is directly available for plant uptake. The microbial activity in the soil mineralizes the organic form of N into mineral forms of N, e.g., NH₄⁺, NO₂⁻, and NO₃⁻.

In some areas, septic tanks are suspected to cause an increase in the content of groundwater nitrogen.

Figure 2.2 Field drains embedded in gravel to collect the surplus drain.

(from Kowalik et al., 1985)
Soil nitrogen in the inorganic form seldom exceeds 1-2% of the total N present (Haynes, 1986), except where large applications of inorganic N fertilisers have been made.

In natural mature ecosystem once a steady state has been reached, the rate of N input is approximately balanced by the N outputs. However product removal and ecological inefficiencies present in the agricultural production systems require large N inputs to balance N losses.

Figure 2.3 illustrates the N cycle within an agricultural production system.

2.3.1 N Mineralisation and Immobilisation

During the decomposition of plant and animal residues organic forms of nitrogen are converted to inorganic forms (e.g. NH$_4^+$) by the process of mineralisation. Microbial activity is the primary facilitator of this although some non-biological processes do occur.

Some of the N present in the residue during this breakdown is converted into inorganic forms and either assimilated into microbial tissues or microbially complexed into soil humus, resistant to further microbial attack. This process is known as immobilisation.

These two processes occur simultaneously in the soil systems, the net effect being determined by the C:N ratio of residues being decomposed. If the C:N ratio is high (>25 :1) net immobilisation usually occurs (Allison, 1973), and with a low C:N ratio e.g. (10:1) net mineralisation occurs (Bartholomew, 1965).

2.3.2 Nitrification

The nitrification of NH$_4^+$ is an important part of the mineralisation process whereby NH$_4^+$ is converted to NO$_2^-$ and NO$_3^-$ following the ammonification of organic N. The process of nitrification can be summarised as follows:

$$\text{Fast} \quad \text{Very fast} \quad \text{NH}_4^+ \rightarrow \text{NO}_2^- \rightarrow \text{NO}_3^- \quad (2.1)$$

The biological oxidation process, mediated by two groups of microbes, is responsible for the conversion of NH$_4^+$ to NO$_3^-$. *Nitrosomonas* is typical of six or so genera which oxidise NH$_4^+$ to NO$_2^-; \textit{Nitrobacter* represents the rather fewer genera which oxidise nitrite to nitrate. Nitrifying bacteria perform a valuable function in that NO$_3^-$-N is more available for plant uptake and thus more effective as a plant nutrient than NH$_4^+$-N. However NH$_4^+$-N is retained well in soils compared to NO$_3^-$-N which is readily leached. Nitrification therefore may result in loss of inorganic N from the rootzone (Postgate, 1978).
Figure 2.3 Nitrogen Cycle in agricultural production system
Rates of nitrification vary according to the factors affecting the biological environment, namely pH, mineral nutrient status, aeration, temperature and moisture. Well aerated, warm, moist soils close to neutral pH favour rapid rates of biological oxidation of NH$_4^+$+. Nitrification rates decrease with depth and in soils below 15% soil moisture.

2.3.3 Denitrification, Nitrification and Cheomodenitrification

Losses

The processes of bacterial denitrification, nitrification and reactions of NO$_2^-$ with the soil components represent possible pathways of loss of gaseous N from the ecosystem. Bacterial denitrification is a biochemical reduction process mediated principally by anaerobic bacteria such as *Pseudomonas* and a few other genera. The process can be summarised as follows:

\[
2\text{NO}_3^- \rightarrow 2\text{NO}_2^- \rightarrow \uparrow \text{2NO} \rightarrow \uparrow \text{N}_2\text{O} \rightarrow \uparrow \text{N}_2
\]  

(2.2)

Non-denitrifying fermentative bacteria and fungi, and autotrophic nitrifying bacteria (e.g. *Nitrosomonas*) may also produce gaseous N products. Denitrifying bacteria are thought to be most important organisms contributing to the losses of nitrogenous gases from soils under anaerobic conditions.

The effect of gaseous N loss via bacterial denitrification and nitrification may vary considerably. Rolston and Broadbent (1977) calculated a loss of 13 kgN/ha (9% of applied fertiliser) from cropped plots over an entire growing season. Rolston et al. (1976) measured gaseous N losses under different moisture, temperature and cover conditions following the application of manure. Up to 75% of N, applied as manure, was lost under wet treatments. In grass plots receiving either 250 or 500 kgN/ha/yr in the form of fertiliser, losses have been estimated at 11 and 29 kg N/ha/yr respectively (Ryden, 1981).

Although only small and variable amounts of data is available, Colbourn and Dowdell (1984) generalised that direct and indirect estimates of N$_2$ plus N$_2$O from soils range from 0-20% of fertiliser N applied to arable soils and 0-7% on grassland soils.

Chemodenitrification occurs when NO$_2^-$ reacts with soil components resulting in a chemical reduction process unassociated with microbial activity forming gases such as N$_2$, N$_2$O and NO. Accumulation of NO$_2^-$, allowing significant rates of chemodenitrification, mainly occurs when nitrogenous fertilisers that form alkaline solutions upon hydrolysis are band applied. Build up of NO$_2^-$ during denitrification of NO$_3^-$-N applied as fertiliser also occurs.
The significance and magnitude of chemodenitrification under field conditions has yet to be established but gaseous losses via this process are not regarded to be large (Haynes, 1986).

**2.3.4 Ammonia Volatilisation**

A significant loss of N in the form of NH$_3$ gas can result from application of ammonium containing fertilisers, urea, or urine-N, which hydrolyse to ammonia. The rate of hydrolysis, and therefore volatilisation, is affected by soil temperature, moisture levels and pH. Haynes and Sherlock (1986) concluded that amounts of NH$_3$ volatilised from fertilisers are variable but loss of fertiliser N applied to the surface of grassland or bare soil could be in the range of 0-25%. Volatilisation from urine patches can be high. Ball and Keeney (1981) measured losses of up to 66% of applied urine-N during warm dry weather with a 28% average calculated over a range of seasonal conditions. Sherwood (1981a) concluded that 40-80% of ammonium nitrogen was lost through ammonia volatilisation within approximately seven days of pig slurry application on grassland.

**2.3.5 Plant Uptake of Inorganic Nitrogen**

Nitrogen plays a central role in plant productivity because it is a major component of amino acids, proteins, nucleic acids and chlorophyll. The predominant form of N available to plants is NO$_3^-$ since under most soil conditions NH$_4^+$-N is rapidly nitrified to NO$_3^-$-N. In addition to increasing plant growth, uptake of nitrate serves to reduce the amount of inorganic soil N which is available for leaching. Changing environmental factors (especially temperature, moisture and soil pH) and different crop types affect the amounts of N removed by the crops. Table 2.3 shows the variation between the crops in terms of total N removal.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Mean N%</th>
<th>Range</th>
<th>Average crop yield (t/ha)</th>
<th>N removal kg/ha</th>
</tr>
</thead>
<tbody>
<tr>
<td>Barley</td>
<td>1.78</td>
<td>1.45-2.35</td>
<td>5 (15%M)</td>
<td>76</td>
</tr>
<tr>
<td>Wheat</td>
<td>2.07</td>
<td>1.65-2.65</td>
<td>5 (15%M)</td>
<td>88</td>
</tr>
<tr>
<td>Oats</td>
<td>2.09</td>
<td>1.90-2.40</td>
<td>5 (13%M)</td>
<td>91</td>
</tr>
<tr>
<td>Peas</td>
<td>3.37</td>
<td>3.02-3.85</td>
<td>4 (14%M)</td>
<td>116</td>
</tr>
<tr>
<td>Soyabean</td>
<td>5.46</td>
<td>4.94-6.07</td>
<td>3 (12%M)</td>
<td>144</td>
</tr>
<tr>
<td>Maize</td>
<td>1.48</td>
<td>0.80-2.00</td>
<td>12 (14%M)</td>
<td>153</td>
</tr>
</tbody>
</table>

M= Moisture content
In conditions unfavourable for nitrification to proceed (e.g. poorly aerated/waterlogged soils), NH$_4^+$-N is the major form available. Ammonium-N is however converted to other forms in the plants itself as a buildup of ammonium in plant cells is toxic.

Quin (1982) estimated plant uptake rates of approximately 160-200 kgN/ha/yr for sheep grazed Canterbury pastures. It should be remembered that the plant uptake of inorganic N will be affected by the factors altering the pasture growth such as soil moisture levels, temperature, sunshine hours, nutrient status of the soil, and will vary seasonally.

Considerable N loss may also occur in grazed pastures. Carran et al. (1982) concluded that herbage uptake of urine-N was only 15 and 22% of applied N in dry and wet treatments respectively. Volatilisation, leaching and fixation processes accounted for the majority of the balance. Pain and Smith (1986) calculated the apparent recoveries of cow slurry N by grass cut for silage (Table 2.4).

**Table 2.4 Apparent Recovery by Grass of N in Cow Slurry**
(from Pain and Smith, 1986)

<table>
<thead>
<tr>
<th>Year</th>
<th>Apparent uptake of slurry N applied (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Early application$^a$</td>
</tr>
<tr>
<td></td>
<td>L</td>
</tr>
<tr>
<td>1980</td>
<td>26</td>
</tr>
<tr>
<td>1981</td>
<td>26</td>
</tr>
<tr>
<td>1982</td>
<td>31</td>
</tr>
<tr>
<td>Mean</td>
<td>28</td>
</tr>
</tbody>
</table>

$^a$ Slurry applied pre-first cut: March-April
$^b$ Slurry applied post-first cut: May-June
L, Low rate of slurry application (80 kg of N per hectare)
H, High rate of slurry application (160 kg of N per hectare)

These values were calculated by comparing N uptake on plots receiving slurry or nitrogen fertiliser with that of those receiving no treatment. Apparent recovery of slurry N was generally of a lower magnitude than the efficiency of utilisation of N in the slurry when compared with the response to fertiliser N (Table 2.5). This is not surprising since recoveries of fertiliser N are usually well below 100%.
Table 2.5 Mean Efficiency of Utilisation of N in Cow Slurry
Compared with that in Mineral Fertiliser
(from Pain and Smith, 1986)

<table>
<thead>
<tr>
<th>Year</th>
<th>Efficiency of slurry N (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Early application&lt;sup&gt;a&lt;/sup&gt;</td>
</tr>
<tr>
<td></td>
<td>L</td>
</tr>
<tr>
<td>1980</td>
<td>34(8)</td>
</tr>
<tr>
<td>1981</td>
<td>42(9)</td>
</tr>
<tr>
<td>1982</td>
<td>37(7)</td>
</tr>
<tr>
<td>Overall mean</td>
<td>38</td>
</tr>
</tbody>
</table>

<sup>a, b, L, H, As Table 2.4<br>\( \), Number of site years</sup>

2.3.6 Adsorption and Fixation Processes

The amounts of inorganic N which a soil can retain against removal by leaching or volatile loss, depends on the form and amount of both the inorganic and organic fractions of the soil and also the type of clay minerals present. Because soil clay and organic matter have a predominantly negative charge they are able to attract and hold positively charged cations such as NH<sub>4</sub><sup>+</sup>-N (Figure 2.4). A factor which often determines N movement in the soil is the degree of interaction between N in the solution and the surfaces of the soil clay and organic matter. Soils having the high cation exchange capacity (cation exchange capacity is represented by the total negative charge on a soil that could hold positively charged ions) can more effectively protect NH<sub>4</sub><sup>+</sup> from leaching and losses of ammonium are only likely to be a problem in soils with extremely low cation exchange capacities (Cameron and Haynes, 1986). Prevention of rapid NO<sub>3</sub><sup>-</sup>-leaching by surface adsorption of NO<sub>3</sub><sup>-</sup> is only possible in soils which have a significant anion exchange capacity. Clay minerals with a 2:1-type structure (e.g. vermiculite) have the ability to fix NH<sub>4</sub><sup>+</sup>-N and K<sup>+</sup> ions within their interlayers (Figure 2.4). Soils rich in these clay minerals fix NH<sub>4</sub><sup>+</sup>-N readily making it unavailable for plant uptake.

The relationship between NH<sub>4</sub><sup>+</sup> in soil solution and fixed NH<sub>4</sub><sup>+</sup> can be summarised as follows: (Nomnik and Vahtras, 1982)

\[
\begin{align*}
\text{Fast} & \quad \text{Solution NH}_4^+ & \quad \text{Exchangeable NH}_4^+ \quad \text{Slow} \\
\text{Intermediate NH}_4^+ & \quad \text{Very} \quad \text{Fixed NH}_4^+ \quad \text{Slow} \quad (2.3)
\end{align*}
\]
Figure 2.4 Ammonium adsorption and fixation processes
A common observation in soils with predominantly 2:1-type clay minerals occurring in temperature regions is that NO$_3^-$ moves freely through soils with rain or irrigation water (Wild and Cameron, 1980a,b). Nitrate is not usually fixed in soil but can be adsorbed in the soils with high anion exchange capacity such as volcanic and/or tropical soils high in allophane, iron and aluminium oxide and hydroxides, and 1:1-clay minerals (e.g. Kaolinite) (Sumner and Reeves, 1966). Nitrate adsorption is a mechanism restricting free movement of NO$_3^-$-N with water under field conditions, however the presence of organic matter can tend to decrease adsorption so that NO$_3^-$-N adsorption is generally greatest in subsoils (Black and Waring, 1976).

2.3.7 Leaching of Inorganic Soil N

The high mobility of NO$_3^-$-N in most soil systems makes the leaching loss of NO$_3^-$-N the most important N loss on cultivated or intensively grazed agricultural lands where exogenous inputs of N are relatively high. Steele (1982) found that the presence of animals on agricultural lands may considerably increase the leaching losses of N than on ungrazed areas. Quin (1982) demonstrated that the leaching loss of NO$_3^-$-N from a urine patch under surface irrigation could be 20% of the urine urea-N (75% of the total urine-N). This amounted to a net loss of approximately 80 kgN/ha. The reported data is variable and the size of the NO$_3^-$-N loss via leaching depends on several factors including soil texture, rainfall, irrigation and those factors that influence the amount of nitrate in the soil.

Quin (1977) estimated that leaching losses under dryland conditions were 15-30 kgN/ha/yr while surface irrigation increased the leaching losses to around 70-130 kgN/ha/yr. Ball (1979) measured leaching losses from grazed pastures in the Manawatu during late autumn, early winter when no fertiliser N was applied, and this amounted in excess of 50 kgN/ha, while leaching losses of about 200 kgN/ha/yr were reported after the application of 450 kg fertiliser N/ha. Adams (1981) measured the leaching losses in Paparua county following spring application of piggery effluent. The study shows that the two main periods of leaching occurred: the first was between October and December when approximately 60 kg/ha of nitrate-N was leached from the bottom of three depth increments (40-100 cm). The main period of leaching occurred in the autumn and winter from April to September by which time nitrate levels were very low. Little movement of nitrate occurred over summer when evapotranspiration exceeds rainfall, although periods of heavy rainfall caused significant downward movement of nitrate, but no leaching occurred below 100 cm. The total amount of nitrate-N leached was approximately 375 kg/ha. Adams (1981) also carried out similar study at a different site in Paparua county where the rate of effluent nitrogen application was approximately 450 kg/ha. It was concluded that about 400 Kg/ha of nitrate-N could have been leached.
Leaching of N from the soil profile is important as it causes a loss or reduced availability of an essential plant nutrient and may result in nitrate pollution of groundwater.

Nitrate contamination of groundwater from animal waste has been linked to both grazing animals and land application of animal waste (Walker and Kroeker, 1982). One study showed that the levels of nitrate leached below grassland grazed by cattle were 5-6 times those leached below comparably cut but ungrazed grassland. The enhanced movement of nitrate below the grazed grassland was attributed primarily to the return in urine and faeces and represented as much as 90% of the nitrogen in the herbage consumed by cattle (Hubbard and Sheridan, 1989). Sherwood (1981a) conducted a study with pig slurry application at three different rates, three times a year and concluded that the leaching losses increased as the rate of slurry application increased. Sherwood (1981a) estimated that the leaching losses accounted for approximately 5% of the total applied N at the lowest rate of the slurry application and approximately 13% at the highest rate.

Agricultural management practices can affect the extent of nitrate leaching. Walker and Kroeker (1982) showed that 50-75% of the annual deep percolation of nitrogen came from preplant or emergence irrigation. Cutting the first irrigation in half reduced the annual nitrogen flux below the 2.0 meter depth by more than 50%. In turfgrass experiments (Synder et al., 1984) fertigation (fertiliser applied by sprinkler irrigation) and irrigation based on soil moisture content produced the lowest nitrate leaching (less than 1% of applied nitrogen). The same study showed that high rainfall shortly after applying water-soluble nitrogen can produce leaching regardless of the irrigation method.

### 2.3.8 Legume N Fixation, Fertiliser Inputs and Animal Returns

Legumes are the primary source of N input in New Zealand pastoral agricultural systems. The legumes utilise the atmospheric N through the symbiotic association existing between N fixing bacteria such as *Rhizobium*, and plant root nodules.

In pastures, legumes are usually grown with companion grass species which exert both positive and negative effects on N fixation. This reduces legume growth but grasses remove mineral N from the soil encouraging N fixation. As soil mineral N levels rise fixation rates generally decrease (Ball, 1979).

Symbiotic N fixation was measured at nine sites throughout N.Z. in a series of experiments by Hogland et al. (1979). The approximate N input over a 2-year period ranged from 380 kg N/ha/yr in Kaikohe to 107 kgN/ha/yr in Gisborne, under a rotational sheep grazing regime. The developed low land pastures fixed 184 kgN/ha/yr on average. The rates of only 34 kgN/ha/yr fixed on low fertility were also found under developed hill country soils.
Non-symbiotic N fixation by free living soil microorganisms occurs but the rates are low compared to the symbiotic N fixation. Ball (1979) measured rates of 10 KgN/ha/yr in grass only plots on a fine sandy loam.

Under pasture systems N fertiliser applications are normally justified only when soil N levels or pasture legume populations are low. However under cropping systems exogenous fertiliser inputs are frequently needed to obtain high yields, especially in the case of non-leguminous crops. Large and frequent application of fertilisers results in high levels of soil inorganic N far in excess of immediate plant requirements, that in turn may increase the risk of N loss via leaching or gaseous emission. Fertilisers which become rapidly plant available (i.e. NO₃⁻-N type such as ammonium nitrate and calcium nitrate) tend to have low recoveries when periods of high water flux occur following fertiliser application.

2.4 The Mechanism of Leaching

Two prerequisites are essential for the process of NO₃⁻ leaching to occur: (a) there must be a mineral N accumulation in the soil and (b) there must be downward movement of water in the soil profile. The extent to which the percolating water mixes with and displaces the soil solution as it moves through the soil also influences the mechanism of NO₃⁻ leaching (Cameron and Scotter, 1986). However if it is assumed that the steady state water conditions exist in a homogeneous soil and that there is no interaction between the NO₃⁻ ion and the soil, then NO₃⁻ movement can be described by a combination of three processes: convection, diffusion and dispersion (Cameron and Haynes, 1986).

2.4.1 Convective Flow

Convective flow or mass flow refers to the solute transport due to mass flow of water alone. The hydraulic gradient is responsible for the water and the solute movement and the rate of movement is dependent on the magnitude of the hydraulic gradient and the hydraulic conductivity of the soil. Equation 2.4 describes such movement

\[
\frac{\partial c}{\partial t} = - U \frac{\partial c}{\partial x}
\]  

(2.4)

where, \( c \) = concentration of NO₃⁻ \((\mu g/ml)\); \( t \) = time (days); \( U \) = average pore velocity \((cm/day)\), obtained by dividing the rate of water flow by the volumetric water content of the soil; and \( x \) = linear distance in the direction of the flow \((cm)\) (Cameron and Haynes, 1986).
When a slug of solute is introduced to the soil surface convective transport would move the solute down the profile by piston displacement (Figure 2.5). In reality however, solute movement is not as simple as this and the band of solute does not remain contiguous but spreads out through the soil profile due to diffusion and hydrodynamic dispersion (Hillel, 1980).

![Figure 2.5 Piston flow: (a) Band of NO₃⁻ applied to soil surface (b) NO₃⁻ band moved distance U by piston displacement resulting from V mm water applied to soil at moisture content θ (Cameron, 1983)](image)

2.4.2 Diffusive Flow

When there is an uneven distribution of solute in the soil solution there is a diffusive flux of solute from the areas of high concentration to areas of low concentration. Movement by diffusion can be described by equation 2.5:

\[
\frac{\partial c}{\partial t} = D_s \times \frac{\partial^2 c}{\partial x^2}
\]  

(2.5)

where, \(D_s\) = effective diffusive coefficient in soil (cm²/day) (Cameron and Haynes, 1986).

In soil, solute diffusion can only occur in the fraction of soil volume occupied by water. The effective diffusion coefficient in a water saturated soil is less than that in bulk water because of (a) the smaller volume of soil solution available for diffusion, and (b) the increased path length due to pore tortuosity (Wild, 1981; Cameron and Haynes, 1986). The diffusion coefficient \(D_s\) of NO₃⁻ in soil at -1.0 kPa is about \(10^{-6}\) cm²/sec (Nye and Tinker, 1977) and an average NO₃⁻ ion would move about 0.5 cm per day.
2.4.3 Hydrodynamic Dispersive Flow

The mechanical action of a solution flowing through the soil causes mixing of the solute and thus further complicates the process of transport. This is usually referred to as hydrodynamic dispersion. This process enhances the dispersive effect of diffusion and often completely masks it. Hydrodynamic dispersion occurs because of (a) the flow velocity within a single pore is not uniform since it is faster at the pore centre, (b) the large variation in pore size within a soil results in an extremely wide range of pore water velocities (Figure 2.6), and (c) the path length of pores fluctuates greatly due to the tortuosity of pore geometry. Figure 2.7 shows the effects of hydrodynamic dispersion on the band of the solute.

![Figure 2.6 Flow velocity gradient within a pore (Cameron, 1983).](image)

![Figure 2.7 Hydrodynamic spread of nitrate band (Cameron, 1983).](image)
2.4.4 Combined Solute Flux

The combined effects of the convective-diffusive-dispersive mechanisms on soil solute flux can be shown by equation 2.6:

\[
\frac{\partial c}{\partial t} = E \frac{\partial^2 c}{\partial x^2} - U \frac{\partial c}{\partial x}
\]  

(2.6)

where, \(E\) = dispersion coefficient, also called as apparent diffusion coefficient, and is the sum of diffusion plus mechanical dispersion.

\(E = D_s + mU\), where \(m\) = dispersivity (Cameron and Haynes, 1986).

The value of \(E\) depends on the flow velocity and tends to increase with increasing values of \(U\) (Nielsen and Biggar, 1963).

2.5 Soil Physical Properties Affecting Water Movement Through the Soil Profile

2.5.1 Soil Texture and Water Storage Capacity

The hydraulic conductivity and water storage capacity are mainly determined by two major factors: (1) soil texture and (2) soil structure.

After rainfall, surface runoff rather than infiltration will occur on soils with low hydraulic conductivity, reducing the opportunity for leaching (Cameron and Scotter, 1986). The initial infiltration rate is influenced by the existing soil water content (Hillel, 1971). Dry soils have a greater matric potential gradient hence a greater initial infiltration rate than wet soils. Infiltration rate declines exponentially with the increase in soil water content of profile, that reflects a reduction in the matric potential gradient. Crust formation following aggregate breakdown, clay swelling and air entrapment are other factors that contribute to this decline (Hillel, 1971).

Leaching losses are also influenced by the soil water storage capacity. A large amount of water is required by a soil with a high available water holding capacity in order to re-wet the root zone after a dry period. In Canterbury most of the autumn and early winter rainfall is used to satisfy the soil moisture deficit and thus does not cause a water and solute loss by through drainage (Cameron and Scotter, 1986).
The depth to which \(\text{NO}_3^-\) is leached approximately equals the excess of rainfall over evaporation, divided by the volumetric water content of the soil at field capacity, assuming movement through the soil is uniform (Cameron and Wild, 1982; Cameron and Scotter, 1986). Therefore for any given water input, nitrate can move two or three times as deep in a coarse sand as in a loam or clay soil due to different water contents at field capacity. Nitrate losses are normally greater from poorly structured sandy soils compared with those from coarsely structured clay soils (Sommerfeldt et al., 1982; Avnimelech and Revah, 1976).

2.5.2 Pore Size - Distribution and Continuity

The rate of water flow through a soil profile is influenced considerably by the pore size diameter. This phenomenon is best illustrated using Poiseuille's Law (Hillel, 1980):

\[
Q = \pi R^4 \Delta p / 8 \eta L
\]

(2.7)

where, \(Q\) is the volume of water flowing through a section of length \(L\) per unit time; \(R\) is the pore radius; \(\Delta p / L\) is the pressure drop per unit distance; and \(\eta\) is the viscosity of the water. This equation shows that the volume flow rate \(Q\) is proportional to the fourth power of the radius \(R\), therefore a pore with an effective radius of 1 mm will conduct a volume of water 10^{12} times greater than a pore with a radius 1 \(\mu\)m.

2.5.2.1 Macropore Movement

Large continuous pores, or a network of large channels, play a very important role in vertical movement of water through soil and consequently the transport of solutes and suspended matters. Nearly 100 years ago Lawes et al. (1882) found that a large part of water added to soil profile moved immediately through open channels and interacted only slightly with the water in the soil itself. Subsequent drainage was found to be more representative of the water in the soil matrix.

Macropores created by earthworms and roots or cracks may have an overriding effect on the infiltration rate and leaching patterns of a soil. This is due to the preferential pathways provided by them for water flow and when this occurs, usually during ponding or near field saturation, the majority of soil water and solutes held in small pores within aggregates are effectively bypassed. This effect results in either a reduction in leaching due to the protection afforded within aggregates (Kanchanasut and Scotter, 1982), or enhanced leaching when applied
fertiliser or organic wastes do not have sufficient time to diffuse into aggregates, before heavy rainfall or irrigation occurs (Cameron and Scotter, 1986).

Beven and Germann (1982) have shown experimentally that Darcian principles of water flow in homogeneous media are not able to give a full description of infiltration and redistribution of water in soils containing macropores. Such theories have been applied to miscible displacement experiments with columns of sand, resin, glass beads or finely sieved and repacked soil. The underlying assumptions are that a reactive solute moves through the media at the same velocity as the water and that all residual water in the column is displaced by the incoming water. As described before, in undisturbed, naturally structured soils (especially with macropores) only partial displacement of resident water and solutes by incoming water and solutes occurs (Thomas et al., 1978; Lawes et al., 1882).

There is no doubt that water will move through large pores under saturated conditions and that they have a very important influence on the saturated hydraulic conductivity of soils, even though they may contribute a very small amount to the total porosity of the soils.

2.5.2.2 Types of Macropores

Macropores are formed in various ways: (a) by shrinkage at natural planes of weakness on drying (Brewer, 1964); (b) chemical weathering leading to solution piping (Reeves, 1980); (c) freeze-thaw cycles; (d) mole draining and sub-soiling (Beven and Germann, 1982); (e) plant roots and (f) soil fauna. Beven and Germann (1982) concluded that the most soils contain some macropores, the nature and volume of which depends on a dynamic balance between constructive and destructive processes.

The shape of macropores varies from planer slits (cracks or fissures) through voids of irregular cross-section (vughs) to cylindrical pipes.

2.5.2.3 Effects of Macropore Flow on Nitrate-Nitrogen Movement Through Undisturbed Field Soil

Plant uptake and potential pollution of surface and subsurface groundwater are very much governed by the water and nutrient movement through field soils. Because of the negative charge associated with the nitrate ion, and the potential harmful effects of high concentration of nitrate in drinking water, attention has been devoted to NO$_3^-$-N movement within the soil profile (Shuford et al., 1977). Movement of water in soil macropores has two important implications to the process of NO$_3^-$ leaching: (1) when the infiltrating water contains a high concentration of NO$_3^-$ then macropore flow will lead to extensive leaching at a faster rate than predicted by equation
(2.6) and illustrated in Figure 2.8, (2) when $\text{NO}_3^{-}$ is present within the micropores of aggregates it may be bypassed by the bulk of flowing water and this leads to solute retention and a slower than predicted rate of leaching (Figure 2.8) (Cameron and Haynes, 1986).

![Figure 2.8 Schematic diagram of macropore bypass and macropore leaching](image)

Shuford et al. (1977) conducted a field study to determine whether large pores represent a major pathway for $\text{NO}_3^{-}$-N movement through undisturbed soil. A solution containing 410 $\mu$g/ml $\text{NO}_3^{-}$-N as KN03 was added to three plots enclosed by wooden planks to prevent runoff. Most of the $\text{NO}_3^{-}$-N added in the solution moved vertically with the water through profile, when initial and final soil samples were taken at various profile depths and analysed. Ion movement was influenced by the large soil pores between structural units. On another plot measurements were taken to compare field measured $\text{NO}_3^{-}$-N with theoretical calculations, using time dependent water flow velocity and dispersion equations. Soil solution samples were taken and analysed for ions. Due to incomplete mixing of water and ions with all soil pores, the theoretical equations did not completely explain the field measured $\text{NO}_3^{-}$-N distributions.

The effect of macropores on nitrate leaching is complex not only because of the uncertainty about the relative contribution of different pathways to water flow but also because of the variation in the nitrate concentration over short distances. Within a soil block of 1 m$^3$, the uneven distribution of N-fertiliser, excreta from animals and of mineral-N derived from the soil organic matter creates spatial heterogeneity (White, 1985). Scientists have suggested that when nitrate is held within soil aggregates, it will be protected from the leaching when bypass flow occurs. They have also observed that if nitrate has recently been applied to the soil, or if soil generated nitrate is held on the outside of the aggregates, then bypassing flow causes it to leach faster than it would by uniform miscible displacement (Cameron and Haynes, 1986). Shaffer et
al. (1979) compared the nitrate concentration of water sampled by suction cups at several depths in an undisturbed soil with that of water draining from macropores at 1.2 m. Water with and without nitrate was applied immediately to the surface. The macropore drainage responded immediately to a change in the concentration of the applied water (increase or decrease), whereas the concentration in the matrix flow showed only a small gradual increase with time. It was estimated that 90% of the total flow in this soil occurred in pores ≥ 150 μm diameter.

2.6 Seasonal and Climatic Effects on N Leaching Losses

2.6.1 The Influence of Rainfall

Major leaching losses of N only occur once soil moisture levels are above field capacity. However losses may also occur due to the flow down the edges of soil macropores when the soil is below field capacity (Williams, 1975). Summer rainfall is generally used to satisfy soil moisture deficits and evapotranspiration losses are usually high enough to prevent soils experiencing significant summer leaching losses in temperate agricultural systems. Once the soil is approaching or has reached field capacity, autumn rainfall can leach any nitrate released by mineralisation or residual fertilizer-N. Cameron and Haynes (1986) concluded that major leaching losses occur in winter because of the large excess of rainfall over evapotranspiration and low uptake of N by crops or pasture. Williams (1975) used a tile drain study in Britain and found increased drainage volume in late autumn when soil field capacity had been reached. The total NO$_3^-$ loss during winter and spring was closely related to the water flow rather than the NO$_3^-$ fluctuations. Figure 2.9 illustrates Williams (1975) results.

2.6.2 Temperature Effects on N Mineralisation

The increase in soil temperature, aeration and moderate soil moisture levels greatly enhance the mineralisation rates (Tham, 1971). Inorganic N buildup is therefore common during summer periods when temperatures are higher and soil moisture levels are moderate to low.

The optimum temperature reported for mineralisation by Tham (1971) was in the range of 37 to 39°C, with 40-100% of the available water holding capacity being regarded as the most favourable soil moisture regime for mineralisation. Field et al. (1985a) calculated leaching losses from rye grass/clover pasture (no tracer added) to be between 60 and 80 KgN/ha. More than 85% of this amount was lost by early August, even though 40% of drainage occurred after this date. Dry summer followed by wet winters can therefore lead to substantial leaching losses of NO$_3^-$.
considerable constraint is placed on production and management techniques to minimise the economic losses and to maintain groundwater quality, if such a marked seasonal pattern of leaching loss occurs.

Figure 2.9 Interactions of season and climate on leaching losses
(Cameron and Haynes, 1986)
2.7 Consequences of N Losses

Losses of N from agricultural lands through runoff of soluble and particulate N as well as NO₃⁻ leaching have several notable consequences. Where losses are large enough to cause a decrease in yield, the economic significance is obvious. However it is the environmental effect of such losses that have recently received more attention (Cameron and Haynes, 1986).

2.7.1 Economic Loss

Leaching losses of NO₃⁻ are not normally large except when fertiliser inputs greatly exceed crop requirements or the ground is left fallow. Leaching losses from intensively grazed pastures may be large, particularly if irrigation follows immediately after the grazing period.

2.7.2 Decreased Soil pH and Base Saturation

Nitrification and the subsequent leaching of NO₃⁻ has an acidifying effect on the surface soil (Wolcott et al., 1965). During the process of nitrification H₃O⁺ ions are released:

\[
\text{NH}_4^+ + 2\text{O}_2 + \text{H}_2\text{O} \rightarrow 2\text{H}_3\text{O}^+ + \text{NO}_3^- \tag{2.8}
\]

Exchangeable cations, displaced by the H₃O⁺ ions, move downward as counterions with NO₃⁻ resulting in a decrease in pH and base saturation of the surface soil (Haynes, 1981a). In fertile soils Ca²⁺ is often the dominant balancing cation for leached NO₃⁻ (Haynes, 1981b).

2.7.3 Eutrophication

Point sources (e.g. sewage outfalls or industrial effluents) or nonpoint sources such as stormwater runoff and leaching and runoff from croplands can generate the nitrate pollution. In terms of N inputs to surface and groundwaters, nonpoint sources makeup by far the largest contribution (Loehr, 1974). It has been indicated by the estimates that more than 90% of the N entering surfacewaters originates from the nonpoint sources and that more than 80% of that portion is from agricultural lands including livestock feedlots. Although point sources of N are of minimal importance on a regional basis they can represent major sources on a local basis (NRC, 1978).
Since P and N are the nutrients limiting the production in most lakes, these nutrients are the most important in stimulating eutrophication (Keeney, 1973). Most low-producing oligotrophic lakes (low in nutrients) are P rather than N limited (Keeney, 1973) due to the paucity of P in the biosphere compared to N. Nitrogen can, however, be a limiting element in some ultraoligotrophic lakes (Forsberg, 1977). The productivity of coastal and estuarine ecosystems is quite often limited by N (Goldman, 1976). In many already eutrophic lakes, biotic productivity is controlled by N because the N/P ratio of pollutants from many sources are far below the ratios required for the plant growth (Cameron and Haynes, 1986). In some oligotrophic lakes where N is the limiting nutrient, the inputs from groundwater, surface runoff, or precipitation may be essential to maintain biological productivity (Keeney, 1982). However over-enrichment of surface waters with nutrients results in a range of changes in water quality that are generally considered undesirable, such as decrease in water clarity, the proliferation of algae and the depletion of dissolved oxygen in the bottom water with concomitant loss of bottom water fisheries.

Eutrophication decreases the recreational value of the lakes through a general loss of aesthetic appeal and it also results in a requirement for increased water treatment before domestic use due to increased colour, taste and odour of the water and its increased chlorine demand. Eutrophication may also result in the partially blockage of irrigation and drainage canals due to excessive growth of aquatic vegetation and in regions can result in increased water loss from irrigation canals because of the evapotranspiration from the floating vegetation (Cameron and Haynes, 1986).

2.7.4 Nitrate Levels in Water

The maximum WHO NO$_3^-$ level in the drinking water is 10 mg/L. The same standards of drinking water have been adopted by New Zealand. It has been established that the young infants (less than 6 months) receiving artificial feed of milk diluted with water containing more than 10-20 mg/L of NO$_3^-$-N may develop methemoglobinemia. This disease that can be fatal, is characterised by the development of a greyish-blue or brownish-blue cyanosis which eventually covers the whole body. It is caused by the nitrates being reduced to nitrites which partially convert the haemoglobin to methemoglobin and this decreases the oxygen-carrying capacity of the blood. More recently, levels of NO$_3^-$-N above the recommended limit have been linked with an increase in the incidence of stomach cancer in the adults, through the formation of carcinogenic nitrosamines (Hill et al., 1973). For several decades the researchers have implied that NO$_3^-$ may affect the cardiac function of the human (Malberg et al., 1978). A relation between the high concentration of NO$_3^-$ in drinking water and hypertension has been recorded (Morton, 1971) but other studies have failed to establish any relationship (Malberg et al., 1978).
2.8 Environmental Impact of Land Application of Wastes

There is no doubt that poorly executed land application of wastes can cause serious environmental pollution. Both nutrient enrichment and bacterial contamination of ground and surface water can occur while spray aerosols can also spread disease organisms (NZAEI, 1984).

Good management of waste application systems will however avert most problems.

2.8.1 Water Pollution

2.8.1.1 Areas Affecting Groundwater

a. Organic Waste and Wastewater. Land application of livestock manure, industrial and municipal sludge, agricultural, industrial and municipal wastewater and other organic wastes is common practice designed to manage and recover fertiliser nutrients and other beneficial elements. Where "disposal" is the primary objective, the potential for contaminating the soil and groundwater is greater. Limiting application rates to the N loading required for the realistic crop yields and selecting and managing crops for maximum nutrient uptake reduces the potential for groundwater contamination.

b. Onsite Sewage Treatment. Potential contamination of groundwater with nitrate-nitrogen, phosphorous and microbiological pathogens is an ongoing concern with onsite sewage treatment systems. Biological organisms and phosphorous can generally be controlled if there is at least one or two meters of unsaturated soil beneath the bottom of the soil absorption system, but even then nitrate may continue to move downward toward groundwater. Site selection must consider soils, hydrology, and depth to water table, and the bottom of the water trenches for the soil absorption system must be kept well above the groundwater level.

Burden (1984) concluded that the irrigation by effluent from meat works on a soil shallow and underlain by a coarse gravel substrata, resulted in subsurface drainage and a measurable impact on the chemical and microbial quality of the groundwater. Nitrate-N concentrations immediately down gradient of the meat works site were raised during the period of effluent irrigation (November-June) but decreased to near background during winter and early autumn. The increase in nitrate content was directly correlated with the drainage from the disposal site. A peak nitrate-N concentration of 12.6 g m\(^{-3}\) corresponded to the period of highest drainage. The World Health Organization has recommended an upper limit of 10 g m\(^{-3}\) nitrate-N
in drinking water (WHO, 1971). Faecal coliform bacteria were also present down-gradient of the disposal sites during the November-June period.

Keeley and Quin (1979) suggested that the chemical composition of the groundwater beneath a site where land application of meat works-fellmongery effluent has been carried out, and where the water-table is 21 m below the ground level, was similar to that of drainage collected from a depth of 6 m. A three depth investigation well situated 500 m downstream from the site showed definite chemical and bacteriological contamination from the effluent irrigation scheme only in the shallowest of the 3 depths.

An outbreak of over 100 cases of gastric disorder in Bramham, England was caused by sewage contamination of a groundwater supply (Short, 1988). Finkelstein et al. (1988) discussed the clinical aspects of responding to an outbreak of 77 confirmed cases of waterborne typhoid fever in Hafia, Israel. In a study cited by Hrudey and Hrudey (1989) occurrence of guinea worm disease \( (Dracunculus medinensis) \) in Nigeria was found to be associated with faecally contaminated water source although this parasite is only transmitted by direct human contact with water, not by faecal contamination. With the application to land of large volumes of minimally pretreated wastewater, considerable potential for adverse health effects exists (Kowal, 1986). These potentials have been briefly summarised by Lance and Gebra (1978) in Table 2.6. They identified the greatest health risks as arising from aerosols in slow rate, groundwater pollution in rapid infiltration and surface water pollution in overland flow. In the USA many states have put blanket bans on the surface discharges to rivers. All wastewater is applied to the land with the result that the non-point-source pollution of streams is now the major concern. Long, slow moving rivers accumulate nutrients as they flow to the coast and the salt concentration gets so high that the water is unsuitable for irrigation (NZAEI, 1984).

2.8.2 Health Risks Associated with Aerosols from Wastes

2.8.2.1 Aerosols and Application of Wastewater

Aerosols are tiny droplets of water, especially common to high pressure system of spray irrigation and carried offsite by wind (Loehr, et al., 1979b).

Aerosols can also be defined as a system of colloidal particles dispersed in gas, smoke or fog; as far as waste water treatment is concerned; aerosols can be created through various processes, especially in activated sludge, trickling filters and spray irrigation. Bacteria, viruses, parasites or chemicals may be contained in aerosols droplets, transportation of these agents in wastewater aerosols is very likely and should be considered a potential source of disease to humans.
Table 2.6 Potential Land Treatment Health Effects (from Lance and Gebra, 1978)

<table>
<thead>
<tr>
<th>Type of land treatment system</th>
<th>Food contamination</th>
<th>Groundwater pollution</th>
<th>Surfacewater pollution</th>
<th>Aerosols</th>
</tr>
</thead>
<tbody>
<tr>
<td>Slow rate</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>++</td>
</tr>
<tr>
<td>Rapid infiltration</td>
<td>-</td>
<td>++</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Overland flow</td>
<td>-</td>
<td>-</td>
<td>++</td>
<td>+</td>
</tr>
</tbody>
</table>

- Little or no potential problem
+ Moderate potential
++ Considerable potential

The dispersion of aerosols in the atmosphere is caused by all surface activities where systems are not closed. Soil injection is an exception. Many agricultural activities may evolve, for example, both pathogenic and nonpathogenic microorganisms. Aerosols vary in size ranging from about 0.01-50 μm (Loehr et al., 1979b) thus making them fairly accessible to intake into the human body. With inhalation the aerosols can enter the body, in fact small aerosol particles will reach the alveoli of lungs. Larger aerosols can be cleared by the respiratory system and be swallowed—infected hepatitis and salmonella disease, for example are transmitted through the gastrointestinal tract. Adsorption of aerosols through skin is also possible and the example of this is contact dermatitis from spraying.

Several field studies have been carried out to measure the emission and airborne spread of viable microorganisms from wastewater collection, treatment and disposal processes. In most of the studies the following basic premises are investigated:

1. Human pathogenic microorganisms are known to be present in the wastewater in large numbers at every stage of handling.
2. Aerosolisation of the liquid wastewater by air-injection, spraying or splashing is mechanically assisted by most modern wastewater treatment processes.
3. Aerosols thus generated very likely contain viable microorganisms, including pathogens.
4. Wind currents carry off airborne microorganisms to a considerable distance.
5. In recent years large populations have inhabited areas close to treatment plants.
6. Airborne pathogens from wastewater may therefore present a health hazard to plant personnel and nearby residents.

7. Therefore measurement of the airborne emission, and dispersion of viable microorganisms by concentration, or both, may provide an indication of an associated health hazard (Hickey and Reist, 1975).

These studies have shown that many types of pathogens and bacteria are emitted at every stage of wastewater treatment and are viable and carried off a considerable distance through wind. However interpretation of the results of these studies in terms of health hazard has been inconclusive, and those, who did make 'health hazard' conclusions, did so by inference. For example, the recovery of index organisms such as coliforms from the air at some distance away from source and in a respirable particle size was believed to be an indication of health hazard from inhalation. Conversely, the failure to recover high concentration of microorganisms, downwind from source, was interpreted to be absence of a significant health risk from the source (Hickey and Reist, 1975).

Teltsch and Katzenelson (1978) carried out a study in which controlled experiments utilizing marker bacteria (a mutant E. coli resistant to the antibiotic nalidixic acid, was added to the wastewater as a marker bacterium) were carried out to evaluate quantitative relationship between enteric bacteria in the effluent used for irrigation and aerosolised bacteria detectable in the air and to evaluate the effect of some meteorological factors such as relative humidity, temperature, wind velocity and solar irradiation on bacterial dispersion in the air. Aerosolised coliforms were detected when their concentration was $10^3$/mL or more in the wastewater. Relative humidity and irradiation appeared to affect the viable bacteria in the air; a positive correlation was found between relative humidity and number of aerosolised bacteria. The correlation between solar irradiation and bacterial level, on the other hand, was negative. During night irrigation, up to 10 times more aerosolised bacteria were detected than with day irrigation. Wind velocity did not play an important role in the survival of aerosolised bacteria.

Katzenelson et al. (1976) compared the incidence of enteric communicable disease in an Agricultural Communal Settlement practicing wastewater spray irrigation with partially treated non-disinfected oxidation pond effluent, and that in other Agricultural Communal Settlement practicing no form of wastewater irrigation. The incidence of *shigellosis salmonellosis*, typhoid fever and infectious hepatitis, was two to four times higher in communities practicing wastewater irrigation. No significant difference was found for the incidence of streptococcal infections, tuberculosis, and laboratory-confirmed cases of influenza.
Moore et al. (1988) identified possible adverse effects on human health from slow rate land application of wastewater. During the first irrigation period at the land treatment system, irrigation wastewater quality approximated that of a low quality primary effluent as determined by microbial and physical-chemical parameters. Seasonal level of human enteric viruses in the wastewater were highest during late summer, coinciding with the time of the substantial crop irrigation.

Feliciano (1979) concluded that there are two schools of thought about the risk of wastewater aerosols. The first states that the risk exists because cases of sickness have been found, but what limits more conclusive evidence is perhaps that the chosen methodologies are not sensitive enough to detect the low concentration of microorganisms in the aerosols.

The other general opinion is that there is no risk of disease from wastewater aerosols, regardless of choice of methodology. If there was a risk of disease, one of the many tests would have indicated something, yet nothing did. Sorber et al. (1984) concluded that in general microbiological aerosols generated in the application of sludge/wastewater do not seem to represent a serious threat to human health for individuals located more than 100 m downwind of the application site. Boutin et al. (1988) found a rather high bacterial viable particle (b.v.p.) count (>2000 b.v.p. per m$^3$ air) at the limits of the slurried areas from land spreading of cattle and pig slurry. No potentially pathogenic bacteria were found. The current spreading practices did not result in a high respiratory hazard levels for people nearby. Shuval et al. (1987) reported on a prospective epidemiological study using carefully screened medical record data from 20 rural collectives in Israel representing a population of over 10,000. They found no excess enteric disease morbidity in the total population exposed to the wastewater irrigation aerosols. An evaluation by Cort et al. (1987) of wastewater irrigation in Monterey county, California involving virological, bacteriological and chemical analyses concluded that treated wastewater irrigation was as safe as well water irrigation.

2.8.2.2 Hazard Reduction

The most favourable conditions for the substantial travel of aerosols occur at night during relatively still conditions, and the simple restriction to day light hours seems likely to decrease the extent of exposure for nearby populations by about a factor of 10. The restrictions for spray irrigation of wastewater should include:

1. Operation during daylight hours only
2. No operation when it is apparent from observing a smoke plume that little turbulent dispersion was occurring
3. Possible restriction to periods when sunlight intensity is above a particular value (Stevenson, 1979).
The use of some form of screening such as shelter belt of trees for aerosol control has been suggested from time-to-time. The capture of aerosols by plant foliage is a complex subject which has been reported at some length by Chamberlain (1975).

2.8.2.3 Disease Organisms

Disease infection is complex and results when the host and the organisms meet under conditions which favour disease (NZAEI, 1984).

Many pathogens must enter through a particular route called the "portal of entry". This differs for different organisms depending on their ability to attack certain organs or parts of the body. Another condition for the disease is that the host must be subjected to an "infective dose". The number of organisms in an infective dose varies with the species of the host and the variety or strain of the micro-organisms.

Many organisms can be transmitted from animals to human (and some vice versa).

- e.g. viral infections
  - cowpox, orf
  - fungal infections
  - ringworm
  - bacterial infections
  - tuberculosis
  - tetanus
  - brucellosis
  - leptospirosis
  - salmonellosis
  - streptococcus
  and,
  - staphylococcus

Pathogens are usually short-lived in the soil environment. Changes in temperature, pH, moisture, and the addition of sunlight restrict their activity outside the host. Hubbel et al. (1973) found a 99% die-off of both coliform and salmonella after about 2 weeks in sandy soil.

Contamination of groundwater from controlled land application processes should be minor, since in the absence of soil fissures water percolation through soil is very effective in removing viruses and bacteria (NZAEI, 1984). McCoy (1969) reported 98% removal in only 350 mm depth of soil.
2.8.2.4 Excreta or Wastewater Disposal

Caulkins et al. (1988) evaluated a combined sewer overflow upgrading project and considered the reduction in incidence of gastroenteritis per 1000 swimmers as a health benefit category. The benefit-cost analysis favoured the sewerage project in 3 of 4 cases. A study cited by Hrudey and Hrudey (1989) for persistence of salmonella enteritidis in drinking and surface waters indicated less than 24 hour survival in organically enriched water but up to 30 days survival in drinking water.

The implication of cryptosporidium as the cause of three waterborne disease outbreak has raised interest in this pathogen’s occurrence in surface waters and its behaviour during wastewater and drinking water treatment (Rose, 1988). The presence of polio virus in Helsinki sewage was confirmed during and after an outbreak of paralytic poliomyelitis (Poyry et al., 1988). A Colorado springs epidemiological study (Schwebach et al., 1988; Durand et al., 1987) evaluated health effects associated with the wastewater irrigation of the recreational parks. Maintaining wastewater faecal coliform levels below 500 per 100 mL was judged to be adequate to prevent any association of park irrigation with incidence of gastrointestinal disease.

2.8.3 Odour and Ammonia Emission

In USA and European countries the number of justifiable complaints about odour from livestock farms have increased in recent years. In UK the total number of premises causing complaint in the year 1984-85 was 2200, over 50% being associated with the pig farms and nearly 50% of the total resulting from the spreading of slurry or manure on land (Pain et al., unpublished paper). A recent survey on the incidence of agricultural odour nuisance (IEHO, 1981) has indicated that land application operations are the most frequent source of complaint, followed by animal housing and slurry/manure storage facilities. Smells can be produced almost continuously e.g. livestock housing, but the most objectionable odours are usually intermittent and arise when manure, which has been stored under anaerobic conditions, are agitated, transported or applied onto the land (Smith and Neilsen, 1983).

In addition to odours, the emission of ammonia from livestock building and from the application of slurry and manure to land is a cause of increasing concern (Voorburg, 1985). Such emissions may not only have an impact on atmospheric chemistry and acid deposition but also represent a decrease in the fertiliser value of the slurry and manure. Quantitative assessment of odour and ammonia emission following the application of slurry to land is a pre-requisite of the development of management guide-lines, machinery, treatment processes etc. for reducing emission.
In a study carried out by Pain et al. (unpublished paper) a system of small wind tunnels was used to collect samples of odourous air and to measure ammonia emission following the application of pig or dairy cow slurry to grassland. Odour strength of air samples were assessed by dynamic dilution olfactometry and the rate of odour emission expressed as odour unit m\(^{-2}\) h\(^{-1}\). For both types of slurry rate of odour emission was high immediately after application but declined to much lower levels within 4-5 hours, and was influenced by wind speed and rate of slurry application. The pattern of ammonia emission was similar to that for odour. Total ammonia loss ranged from 23-76% of the NH\(_4\)\(^+\)-N in cattle slurry, 80% of the total loss occurring within 2-days of application. Loss of ammonia from pig slurry was directly proportional to rate of application but the rate of emission was much lower than for cattle slurry. Strong correlations were established for odour and ammonia emission for both types of slurry.

2.9 Effects of Land Application of Wastes on Soil Properties

Some researchers believe that the land application of wastewater improves a soil's hydraulic properties (Mathers and Stewart, 1980; Mathers et al., 1977). Others have found that manure slurries plug the soil and alter the hydraulic properties (Barringston and Jutras, 1983).

Mathers and Stewart (1980) indicated that the application of feedlot waste increased the soil organic matter and hydraulic conductivity, but decreased the bulk density. In another study Mathers et al. (1977) concluded that time required for the irrigation water to advance in a graded furrow was increased by waste treatment thus, the water intake was greater on waste-treated plots. They also concluded that in the first year the application rates had little effect on irrigation water's advance rate. However water advance rate decreased as the waste application rates increased in the second and third years. Thus, approximately one year was needed before waste improved the soil physical characteristics. This is contradicted by Meek and Oster (1983) who demonstrated that the effects of manure on water intake rates lasted only 1 year after the year of application. However, the two studies did agree that the increased infiltration is generally attributed to an increase in organic matter. Clanton and Slack (1987) concluded that for clay loam and silt loam, saturated conductivities of the soil was less than the saturated conductivities of the wastewater seal. This indicated that saturated conductivity of the seal has a minimum value of approximately 0.28 cm/h or the seal was surficial and separated with water application. For loamy sand, an intermediate seal was formed due to the application of the wastewater. The saturated conductivity continued to decrease for the 12-wk period during wastewater application. When wastewater application was terminated, an immediate increase in the saturated conductivity occurred. This indicated that the sealing is affected by the characteristics of the wastewater during each application rather than a buildup of organic matter on the surface.
Goda et al. (1986) concluded that the repeated application of sewage sludge with more than 3-4 times or the total amount exceeding 30-50 t/ha caused the unfavourable soil pH (>7.5 or <6); decreased the number of soil microbes; reduced crop yields and caused abnormal symptoms in some kinds of crops. Ross et al. (1978) indicated that the irrigation of cores of 10 soils, sampled under introduced pasture, with secondary treated effluent for 16 months had no consistent effect, compared with water treatment, on oxygen intakes, carbon dioxide production or invertase, amalase, urease, phosphatase and sulphatase activities. These biochemical activities, with the exception of phosphatase and sulphatase, were generally significantly correlated with soil organic matter content.

2.10 Social and Cultural Impact of Land Application of Waste: The Maori Perspective

To the Maori, water is essential ingrediënt of life, a priceless treasure left by ancestors for the life-sustaining use of their descendants. The descendants are, in turn, charged with a major stewardship duty, KAITIAKI, to ensure that the treasures are passed on in as good a state, or indeed better, to those following. Water and associated resources confirm life to man, and thereby form a basis for his identification, his belongings, his mana (Taylor and Patrick, 1987).

Maori life was situated and moulded around valued resources, their availability and sustainability. Marae were sited in prime locations for water source and food gathering purposes. In an (historically) hand-to-mouth society such as this, therefore, it is difficult to consider anything other than a conservation ethic, willful pollution or destruction of a waterway or a food resource would probably have an immediate, significantly detrimental effect on the community as a whole (Patrick, 1987).

Water therefore acquired a spirit, a wairua, consistent with how the Maori perceived its quality, and its use, a form of Maori classification system in fact was used, in which the interwoven nature of the spiritual and the physical becomes apparent.

Within such a belief, it is very apparent that the mixing of human waste with natural water is in fact a grievous wrong-doing, an act which would seriously diminish the life-force (mauri) of the water, demean its wairua, and thereby affect the mana, the prestige of those who use it and its resources. Consequently, the Maori question the basic tenet of "disposal of waste to water is an appropriate and valid use of that water" (Taylor and Patrick, 1987).
CHAPTER 3
MATERIALS AND METHODS

3.1 Site and Soil Characteristics

3.1.1 Location And Soil Type

The site of the case study is located on the alluvial plains of mid-Canterbury between the Waimakariri and Selwyn rivers. (See plate 3.1 for the aerial view of the site). The area is flat with an average slope of 5 m per kilometre. The site is about 65 m above sea level. The soil type is Lismore stony silt loam. These soils are formed from the Pleistocene river fans which consist of greywacke gravels in a matrix of sand and silt. It is flat, somewhat excessively drained soil with severe limitations for mixed farming. The stony subsoil is subjected to summer drought and soil is liable to wind erosion. The top soil is described as very dark greyish brown silt loam with stones, friable, moderately developed fine nutty and granular structure, with many roots (Kear et al., 1967) (see Appendix A for full profile description).

3.1.2 Climate

The rainfall is spread evenly throughout the year with a mean annual rainfall of about 700 mm. Significant moisture loss is caused by North-westerly winds, often gusty, strong, warm and dry, whilst the north-easterlies are more dominant and are generally cooler and more consistently strong. South-westerlies are more frequent in the winter and usually accompanied with cold rain. Potential evaporation exceeds monthly average rainfall for up to six months of the year. Pan evaporation shows considerable variations with peaks coinciding with strong north-westerly winds. Average rainfall and evapotranspiration data are presented in Chapter 4. (See Appendix B for rainfall at Burnham sewage plant and Appendix C for Evapotranspiration at Lincoln).
Plate 3.1 An aerial view of the case study site.

Plate 3.2 Measurement of rate and depth of application by the travelling irrigator.
3.1.3 Groundwater

Geographically the plains between Waimakariri and Rakaia rivers may be subdivided into two hydrogeological regions: (a) the unconfined region and (b) the coastal confined region. The site is situated above the unconfined aquifer. The water table depth ranges from 6 to 20 m below ground level (NCCB, 1983).

3.2 Site Measurements

3.2.1 Measurement of Depth and Rate of Application

A traveller irrigator (Briggs Model 80) was used on the site to apply the liquid piggery slurry on farm. To measure the depth of application of the slurry, sixteen raingauges were placed below the irrigator, within the maximum diameter of influence of irrigator, and the slurry was pumped to the irrigator (Plate 3.2). The irrigator was allowed to pass over each raingauge. The area of each raingauge was calculated and the volume of slurry received in each raingauge was divided by the area to give an equivalent depth of application. Two raingauges were placed beyond the diameter of influence of the irrigator to collect the rainfall during the application. The volume of rainfall received was subtracted from the volume of slurry received in the raingauges below the irrigator. The speed of the irrigator was also measured and the time calculated to pass all sixteen raingauges placed below the irrigator. The depth of application was then divided by this time interval to give the rate of application.

3.2.2 Measurement of Soil Bulk Density and Porosity

Soil samples for the determination of bulk density and porosity were taken at 7 places on site using a 50 mm internal diameter soil corer. Samples were taken in 50 mm depth increments to a maximum depth of 200 mm. Samples were stored in air-tight plastic containers and transported back to the laboratory. Bulk density ($\rho_b$) was calculated as:

$$\rho_b = \frac{M_s}{V} \quad (3.1)$$

where $M_s$ is the oven-dry soil mass ($105^\circ C, 24 hr$), and $V$ is the volume of the soil sample.
Porosity $e$ was calculated as:

$$e = 1 - \frac{\rho_l}{\rho_p}$$  \hfill (3.2)

where $\rho_p$ is the particle density (assumed to be 2.65 g/cm$^3$).

### 3.3 Laboratory Analysis of Slurry

Slurry samples were analysed for Total-Nitrogen using the procedure of Bremner and Mulvaney (1982).

### 3.4 Lysimeter Sampling, Preparation and Leaching

#### 3.4.1 Field sampling

A total of 14 undisturbed soil lysimeters were collected from the representative areas which receive piggery waste. Water was applied to the sampling area (c. 1*2 m) in order to bring the soil depth (250 mm) to approximately 'field capacity'. Lysimeter casings (180 mm dia * 250 mm deep) constructed of PVC Stormwater pipe (Garnite 200 Stormwater NZS 7649 A.H.I. Plastic Extrusion Company, N.Z.) with a PVC internal bevelled cutting ring were then placed on the soil surface (Figure 3.1). The lysimeter casings were carefully pushed over prepared sections of soil monolith by applying a minimum possible pressure at the top of casings. To minimize the pressure required to push each casing into the soil, a trench was dug around the casing walls, pressure then applied, and the procedure repeated until the desired depth was reached. Because of the very stony nature of the soil, it was not possible to obtain undisturbed soil lysimeters without the casing hitting a stone. If the stone was not very large, it was removed gently from below the cutting edge and the gap thus generated filled with vaseline and lysimeter casing then pushed gently further down. If the stone below cutting edge was very large, the sample was abandoned and a new site chosen (Plate 3.3).

Once at full depth, liquified vaseline was injected around the edge of lysimeter to eliminate edge flow during subsequent leaching (Plate 3.4). The liquified vaseline did not penetrate into soil (Plate 3.5).

Following injection of vaseline the undisturbed soil within each casing was detached from the underlying soil by pushing a thin stainless steel plate, with a sharp edge, under the
casing. After removal the lysimeters were secured with lids (plus packing) (Plate 3.6) and carefully transported back to the laboratory.

![Diagram of PVC lysimeter dimensions](image)

Figure 3.1 Dimensions of PVC lysimeter used for the case study.

(From Deane, 1988)
Plate 3.3 Insertion of PVC lysimeter into undisturbed field soil.

Plate 3.4 Injection of molten petroleum jelly to prevent edgeflow.
Plate 3.5 Soil core on removal from lysimeter.

Plate 3.6 Cores in preparation for transport back to the laboratory.
3.4.2 Lysimeter Preparation

Surface vegetation was removed from each lysimeter with scissors and a vacuum cleaner. In order to remove the effects of soil smearing, the base of the lysimeter was 'peeled' with a 25 per-cent (w/v) cellulose acetate/acetone mixture. This mixture was applied to the basal soil surface of each lysimeter and then allowed to dry for 8-10 hours (Plate 3.7). When peeled from the soil surface the hardened mixture removed a thin layer of soil and exposed the pores which have been previously sealed by smearing (Plate 3.8).

Depressions in the peeled basal surface of the lysimeters were filled with acid washed sand. The base was then covered with a fine nylon gauze and secured.

3.4.3 Leaching Procedure

Before leaching, the depth of soil column in each lysimeter was measured and the pore volume calculated. The pore volume represents the water and air fraction of total soil volume and was calculated by multiplying total soil volume by soil porosity (calculated by equation 3.2). Before starting the leaching each lysimeter was placed in a water bath and de-aired water poured into the bath until it was level with the surface of the soil. Once the cores were completely saturated (as indicated by ponding in microdepressions on the soil surface), they were carefully taken out of bath, weighed and placed on a table. The lysimeters were left to drain for 24 hours, by which time rapid drainage had ceased. At equilibrium it was assumed that the cores were at approximately 'field capacity'. They were then re-weighed.

Two leaching experiments were chosen because ammonium in the slurry will not leach immediately after the slurry application. This ammonium-nitrate will be converted into nitrate-nitrogen and then be leached. It was assumed that only 10 percent ammonium would convert into nitrate within the study period, after the slurry application, and hence standard solution experiment was conducted with the N concentration one tenth of that measured in the slurry to see the leaching characteristics of nitrate.

3.4.3.1 Standard Solution Leaching Experiment

A standard nutrient solution was prepared with the N concentration one tenth of that measured in the slurry (2000 μg/ml). Calcium nitrate (Ca(NO₃)₂·4H₂O) was used as the nitrogen source in the solution and the weight of calcium nitrate calculated to give a solution of 200 μgN/ml. The volume (ml) of the standard solution was also calculated to give the maximum depth of application of the slurry as on actual site.
Plate 3.7 Applying Cellulose acetate to the base of the cores.

Plate 3.8 Peeling off basal soil surface layer.
Up to 4 cores were run at a time with the cores sitting on the wire grids, secured inside plastic funnels (Plate 3.9). Seven lysimeters were used for standard solution leaching. The calcium nitrate solution was applied evenly over the soil surface using a Mariotte device. The depth of solution applied was equivalent to the maximum depth of slurry application used on the farm. After the standard solution had infiltrated into the soil a nutrient free solution (de-ionised water) was applied to the lysimeters, until approximately two pore volumes of water had been collected from each lysimeter (a time period of 3-8 hours in case of the standard nutrient solution).

3.4.3.2 Liquid Piggery Slurry Leaching Experiment

Five lysimeters were used for liquid piggery slurry leaching. The slurry was applied in the same manner as the standard nutrient solution (see Section 3.4.3.1). Nutrient free water was also applied in the same manner as described in section 3.4.3.1. Leachate was collected over a period of 5-32 days, depending on the individual lysimeter.

Plate 3.9 Apparatus used for leaching studies on undisturbed soil cores showing leachate collection.
3.4.3.3 Leaching of Control Lysimeters

Two lysimeters were used for leaching as a control to monitor background levels of nitrogen leached from the soil.

3.4.4 Leachate Collection and Storage

Leachate was collected at 0.1 pore volume intervals. All samples were stored in tightly stopped plastic bottles and refrigerated until chemical analysis was performed (see Section 3.4.6).

3.4.5 Soil Sampling

Following the completion of leaching duplicate soil samples were collected in 50 mm depth increments from each lysimeter. Again because of the stony nature of soil, a soil corer could not be used for sampling. The soil cores were pushed out from lysimeter casings, divided into four sections longitudinally and soil samples taken from two sections at 50 mm depth increments.

3.4.6 Leachate and Soil Analysis

The initial standard nutrient solution and the leachate were analysed for NH$_4^+$-N and NO$_3^-$-N using the "Standard auto-analyser" procedures. Nitrate-nitrogen was analysed by a modified method of Grasshoff (1969) in which NO$_3^-$-N is reduced to nitrite in a copper/cadmium column which then reacts with sulphanilamide under acidic conditions to form a diazonium salt. This salt then couples with N-1-naphthylene diamine dihydrochloride (NEDD) to form a reddish-purple azo compound, the absorbance of which is read at 530 nm.

Leachate ammonium concentrations were determined by a modified method of Weatherburn (1967), in which NH$_4^+$-N reacts with phenol and alkaline hypochlorite to form indophenol blue. The colour is intensified by the addition of sodium nitroprusside and the absorbance is read at 630 nm.

Soil samples were extracted in a 'field-moist' condition to ascertain the amounts of residual NO$_3^-$-N and NH$_4^+$-N present in the cores after leaching was completed. Samples were extracted using 2M KCL solution (1:2 soil solution ratio) by shaking end-over-end for 25 minutes. Samples were then centrifuged (2200 r.p.m.) for 10 minutes and the supernatant was analysed for NO$_3^-$-N and NH$_4^+$-N using the "Auto-analyser" procedures described earlier.
CHAPTER 4
RESULTS

4.1 Depth and Rate of Application

The maximum depth of application of slurry by the travelling irrigator was calculated to be 35 mm. The maximum rate of application of slurry by the travelling irrigator is c.13.8 mm/hr.

4.2 Total-Nitrogen Concentration in Slurry

Results from the Bremner and Mulvaney (1982) analysis of slurry gave the Total-nitrogen concentration in the slurry to be 2000 μgN/ml.

4.3 Soil Physical Conditions

Results from the field measurement of soil dry bulk density and porosity are presented in Table 4.1.

Table 4.1 Dry bulk density and Porosity of soil of the case study site.
(standard errors for bulk density and porosity in brackets)

<table>
<thead>
<tr>
<th>Depth (mm)</th>
<th>Dry bulk density (g/cm³)</th>
<th>Porosity (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-50</td>
<td>1.49 (+0.08)</td>
<td>43.8 (+0.08)</td>
</tr>
<tr>
<td>50-100</td>
<td>1.78 (+0.12)</td>
<td>34.7 (+0.12)</td>
</tr>
<tr>
<td>100-150</td>
<td>1.66 (+0.13)</td>
<td>37.4 (+0.13)</td>
</tr>
<tr>
<td>150-200</td>
<td>1.60 (+0.03)</td>
<td>39.7 (+0.03)</td>
</tr>
<tr>
<td>Average</td>
<td></td>
<td>38.9 (+1.67)</td>
</tr>
</tbody>
</table>
4.4 Rainfall/Meteorological Data

Table 4.2 The average rainfall and evapotranspiration for 21 years at Burnham over the months

<table>
<thead>
<tr>
<th></th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rain</td>
<td>50</td>
<td>43</td>
<td>56</td>
<td>55</td>
<td>55</td>
<td>59</td>
<td>63</td>
<td>61</td>
<td>40</td>
<td>51</td>
<td>51</td>
<td>53</td>
</tr>
<tr>
<td>ET</td>
<td>167</td>
<td>132</td>
<td>90</td>
<td>53</td>
<td>30</td>
<td>15</td>
<td>16</td>
<td>26</td>
<td>58</td>
<td>85</td>
<td>113</td>
<td>151</td>
</tr>
</tbody>
</table>

(See Appendix B and C for full data)

4.5 Nitrogen Breakthrough Curves

Analysis of leachates from the standard nutrient solution indicated that the nitrogen present in the leachate was in the form of NO₃⁻-N. Whereas the analysis of leachates from the slurry showed that the nitrogen present was in the form of NH₄⁺-N.

Breakthrough curves are presented as the mean concentration of NO₃⁻-N in the standard solution leachate versus pore volume (Figure 4.1); the mean concentration of NH₄⁺-N in the slurry leachate versus pore volume (Figure 4.2); as the mean of the ratio of the concentration of NO₃⁻-N in the standard solution leachate, C, to the initial concentration of NO₃⁻-N in the applied standard solution, Cₒ, versus pore volume (called the 'mean relative concentration') (Figure 4.3); and as the mean of the ratio of the concentration of NH₄⁺-N in slurry leachate, C, to the initial concentration of NH₄⁺-N in the applied slurry, Cₒ, versus pore volume (the mean relative concentration) (Figure 4.4).

Values used to construct the breakthrough curves for both treatments have had the control NO₃⁻-N concentration subtracted from them to eliminate the effects of background NO₃⁻-N levels on the results. The control NH₄⁺-N concentration was negligible.

4.5.1 Mean Concentration of NO₃⁻-N in the Standard Solution Experiment

Figure 4.1 shows the mean concentration of NO₃⁻-N in the standard solution leachate collected at every 0.1 pore volume, with extreme (minimum and maximum) values of NO₃⁻-N concentration in the total number of replicates. The mean leachate concentration was very high (c. 58 µgN/ml) at 0.1 pore volume; peaked at 0.3 pore volume (c.126 µgN/ml) and then decreased
rapidly up to 1.3 pore volume and thereafter followed a very even decline. Individual lysimeters displayed a wide variation in the concentrations at every pore volume illustrating the effect of natural soil variability on solute transport. The maximum peak concentration occurred at 0.5 pore volume in one of the lysimeters. (See Appendix D for individual lysimeter N concentration).

4.5.2 Mean Concentration of NH$_4^+$-N in the Slurry Experiment

Figure 4.2 depicts the mean concentration of NH$_4^+$-N in the slurry leachate with minimum and maximum concentrations of NH$_4^+$-N at every 0.1 pore volume. The mean leachate concentration peaked at 0.9 pore volume (approximately 35 $\mu$g/ml) and after the 1.7 pore volumes it followed a very even pattern but was generally declining. The mean nitrogen concentration of the leachate was usually always greater than 10 $\mu$gN/ml. Even after the slurry was leached by an amount of water equivalent to the amount of rainfall, occurring over the winter period (See Section 4.4). Individual lysimeters also showed a wide variation in the concentration at different pore volumes. The maximum peak concentration (71 $\mu$gN/ml) occurred at 0.8 pore volume. Appendix E shows that NH$_4^+$-N concentration peaked at different pore volumes in individual lysimeters. One of the lysimeters (core 3) had the peak concentration occur at the 0.1 pore volume. Core 4 had two peaks at two different pore volumes.

4.5.3 Mean Relative Concentration of NO$_3^-$-N in the Standard Solution Experiment

The mean relative NO$_3^-$ concentration in the standard solution experiment, versus pore volume, with the minimum and maximum mean relative concentrations measured in any replicates, is shown in Figure 4.3. The peak occurred at 0.3 pore volume and then there was a rapid decrease followed by an even decline in the mean relative concentration of NO$_3^-$-N. (See Appendix F for individual core results).

4.5.4 Mean Relative Concentration of NH$_4^+$-N in the Slurry Experiment

Figure 4.4 shows the mean relative concentration of NH$_4^+$ in the leachate, versus the pore volume, with maximum and minimum mean relative concentrations measured in any replicates. The peak mean relative concentration occurred at 0.9 pore volume and thereafter the curve followed an even pattern and generally declined (See Appendix G for the individual core results).
Figure 4.1 Mean concentration of nitrate in the standard solution experiment
Figure 4.2 Mean concentration of ammonium in the slurry experiment

![Figure 4.2](image-url)

- **Conc. (μg/ml)**
- **Pore volume**

- AM curve
- Minimum
- Maximum
Figure 4.3  Mean relative concentration of nitrate in the standard solution experiment
Figure 4.4 Mean relative concentration of ammonium in the slurry experiment
4.6 Total N Recovery and Loss

Figure 4.5 shows the native extractable-N (mg/layer) in the total depth of soil (See Appendix H for the individual core extractable-N results). The amount of native extractable-N was subtracted from the values of soil extractable-N in both the standard solution and slurry experiments. The native extractable N levels decreased down the profile especially for NO$_3^-$, The proportion of the extractable ammonium and extractable nitrate in the total extractable Nitrogen was almost similar in the top two layers but it decreased for extractable nitrate in the bottom layer of the soil profile.

Total N recovery curves are presented as total mean percent recovery of N in the standard solution experiment (Figure 4.6); total mean percent recovery of N in the slurry experiment (Figure 4.7); total mean extractable-N in soil in the slurry experiment (Figure 4.8); total drainage N loss (kgN/ha) in the standard solution experiment (Figure 4.9); and total drainage N loss (kgN/ha) in the slurry experiment (Figure 4.10).

4.6.1 Total Mean Leachate Recovery

4.6.1.1 Total Mean Percent Recovery of N in the standard solution Experiment

Figure 4.6 shows the total mean percent recovery of N in the standard solution experiment. Initial recovery was very quick and about 80% of N applied was recovered by 1.0 pore volume. Rate of recovery then decreased to a constant rate. Total recovery of N in the leachate equalled 102% following the application of an amount of water equivalent to the amount of rainfall that would occur during an average winter period.

4.6.1.2 Total Mean Percent Recovery of N in the Slurry Experiment

Figure 4.7 depicts the total mean percent recovery of N in the slurry experiment. The percent recovery was fairly even distributed and about 35% of the total N in the leachate was recovered by 1.0 pore volume. The total recovery of N in the leachate equalled about 7.5% following the application of an amount of water equivalent to the amount of rainfall that occurred during the period.
Figure 4.5 Mean native extractable-N in the soil profile

Soil depth (cm)

0-5

5-10

10-15

Extractable ammonium

Extractable nitrate

Min. extractable N

Max. extractable N

(mg/layer)
Figure 4.6 Total mean percent recovery of N in the standard solution experiment

Percentage recovery

Pore volume

Nitrate conc.
Figure 4.7 Total mean percent recovery of N in the slurry experiment
4.6.2 Total Mean Extractable-N in Soil in the Slurry Experiment

Figure 4.8 shows the total mean extractable-N recovered from the soil at different depths with minimum and maximum extractable-N in any replicate. Most of the extractable-N was recovered from the top 50 mm soil. Soil extraction for NO$_3^-$-N and NH$_4^+$-N content found that the NO$_3^-$ levels in different soil layers were low (between 55 and 128 mg or 17% and 29% of total soil Nitrogen per layer). The major proportion of the total soil nitrogen present was in the form of ammonical N (between 71% and 82% of the total soil Nitrogen per layer). Peak soil NH$_4^+$-N levels were recorded at the top 50 mm soil layer. (See Appendix I for individual core results).

4.7 Drainage Loss of N

4.7.1 Drainage Loss of N in the Standard Solution Experiment

Figure 4.9 shows the drainage loss of N in the standard solution experiment. A total of 73 kgN/ha was lost after applying 71 kgN/ha in standard solution. The drainage loss of N was rapid and high upto about 1.0 pore volume and about 80 % of total drainage N loss was lost by this time. After 1.0 pore volume the drainage loss was less rapid.

4.7.2 Drainage Loss of N in the Slurry Experiment

Figure 4.10 shows the drainage loss of N in the slurry experiment. A total of 53.2 kgN/ha was lost in the leachate from the 710 kgN/ha application as slurry. The drainage loss of N in this case was evenly distributed and about 35 % of total drainage loss occurred by 1.0 pore volume.

Table 4.3 shows the amount of N recovered (kgN/ha) after the standard solution and slurry application.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Soil (kg/ha)</th>
<th>Leachate (kg/ha)</th>
<th>Total (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Slurry</td>
<td>544</td>
<td>53.2</td>
<td>597.2</td>
</tr>
<tr>
<td>Standard solution</td>
<td>0</td>
<td>73.0</td>
<td>73.0</td>
</tr>
</tbody>
</table>
Figure 4.8 Total mean extractable-N in the soil in the slurry experiment

Soil depth

- 0-5
- 5-10
- 10-15
- 15-20

Extractable ammonium
Extractable nitrate

Min. extractable N
Max. extractable N
Figure 4.9 Total mean drainage loss of N in the standard solution experiment
Figure 4.10 Total mean drainage loss of N in the slurry experiment
CHAPTER 5
DISCUSSION

5.1 Breakthrough Curves of NO$_3^-$-N and NH$_4^+$-N

5.1.1 The Effect of Preferential Flow

The breakthrough curves obtained from both the standard solution experiment (Figure 4.1) and the slurry experiment (Figure 4.2) were markedly different to those reported from experiments involving piston-displacement or convective-dispersive flow through re-packed soil columns (Figure 5.1). In re-packed soil columns however, the 'in situ' field structure of soil is destroyed. Uniform movement of water and solutes through the soil matrix can then occur, causing complete displacement of initial water by the leaching solution. The peak concentration of the surface applied solute in the leachate occurs at 1.0 pore volume of drainage.

The preferential flow of water and solutes through macropores occurs when either: (1) the soil becomes saturated allowing water to enter the macropores, or (2) vertical flow velocities are such as to exceed the infiltration capacity of the micropores at the surface or at a permeability break within the soil (Germann and Beven, 1981).

Although macropores comprise only 0.1-5% of the total soil volume (White, 1985) they have a profound influence on the water and solute movement within the profile. Table 5.1 shows the hypothetical flow rates of individual cylindrical macropores and their effect on potential soil infiltration rates. (Flow rates were calculated using Poiseuille’s law, see Section 2.5.2).

<table>
<thead>
<tr>
<th>Pore Diameter (µm)</th>
<th>Drainage Tension (-kPa)</th>
<th>Flow rate in one pore ($m^3/s$)</th>
<th>Potential infiltration (mm/hr) into a soil with a macropore area of 1%</th>
<th>5%</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>0.15</td>
<td>$3.8 \times 10^{-6}$</td>
<td>$4.4 \times 10^4$</td>
<td>$2.2 \times 10^5$</td>
</tr>
<tr>
<td>1000</td>
<td>0.3</td>
<td>$2.4 \times 10^{-7}$</td>
<td>$1.1 \times 10^4$</td>
<td>$5.5 \times 10^4$</td>
</tr>
<tr>
<td>200</td>
<td>1.5</td>
<td>$3.8 \times 10^{-10}$</td>
<td>$4.4 \times 10^2$</td>
<td>$2.2 \times 10^3$</td>
</tr>
<tr>
<td>20</td>
<td>15.0</td>
<td>$3.8 \times 10^{-14}$</td>
<td>$4.4 \times 10^0$</td>
<td>$2.2 \times 10^1$</td>
</tr>
</tbody>
</table>

It is obvious from Table 5.1 that the flow rates in macropores are orders of magnitude greater than flow rates within micropores of 20 µm diameter or less.
Figure 5.1 Hypothetical breakthrough curves following the application of a tracer pulse to the surface of a soil column and leaching: (a) piston flow; (b) convective-dispersive flow; (c) preferential flow

5.1.1.1 Standard Solution Experiment: Nitrate Breakthrough Curve

The emergence of the peak $\text{NO}_3^-$-N concentration before 1.0 pore volume of drainage (Figure 4.1) indicated that the water and solutes were flowing through soil macropores, such as earthworm burrows, or even just large inter-aggregate spaces. The shape of the curve in Figure 4.1 shows that extensive preferential flow through macropores had occurred during leaching. A very high mean concentration of approximately 58 $\mu$gN/ml was recorded immediately after drainage had started (0.1 pore volume). This shows that macropore movement can result in a high concentration of nitrate being leached from the soil after only a small amount of net water input and drainage. The peak mean concentration (126 $\mu$gN/ml) occurred at 0.3 pore volume, well before that expected under conditions of uniform miscible displacement of the solute.

The results from this standard solution experiment demonstrate that nitrate released from slurry applied at this site will be readily leached through this soil when drainage occurs. Similar rapid rates of solute leaching have been observed by other workers using undisturbed soil lysimeters (e.g. Monaghan, (1987); and Cameron et al., (1989)).
5.1.1.2 Slurry Experiment: Ammonium Breakthrough Curve

The shape of the NH$_4^+$-N breakthrough curve obtained from the slurry experiment (Figure 4.2) was different from the shape of the curve obtained from the standard solution experiment (Figure 4.1). The mean peak concentration (35 µgN/ml) occurred just before 1.0 pore volume of drainage. The breakthrough curve is also broader than in the standard solution experiment. These differences may indicate that matric flow rather than macropore flow was dominant. This may have been due to the macropores becoming blocked by the thick slurry. It took a time-period of between 5-32 days to conduct the slurry experiment, as opposed to 3-8 hours in case of the standard solution experiment and this lower hydraulic conductivity was attributed to the plugging of macropores by the slurry. The mean nitrogen concentration of the leachate was generally always greater than 10 µgN/ml. The maximum nitrogen concentration obtained from one of the cores was as high as 71 µgN/ml at 0.8 pore volume and even after the total amount of leaching (3.3 pore volumes) the concentration was still higher than 10 µgN/ml. If all of this ammonium was converted to nitrate as it could be, then the nitrate level of water moving into the aquifer would be in excess of the World Health Organisation limits of 10 µgN/ml (WHO, 1971) for drinking water. These results clearly indicate the serious risk of groundwater pollution after the application of the slurry on this site.

These results also emphasise the importance of using undisturbed soil lysimeters when studying solute transport in the field as most of the agriculturally important field soils contain macropores which influence water and solute transport.

5.2 Total N Recovery

5.2.1 Leachate N Recovery

5.2.1.1 Mean Leachate N Recovery in the Standard Solution Experiment

The rapid mean initial recovery of applied NO$_3^-$-N in the standard solution experiment (Figure 4.6) can be attributed to preferential flow through the soil. Approximately 55% of the N applied was recovered within 0.5 pore volume (i.e. an equivalent drainage of approximately 40 mm). About 80% of the N applied was recovered within 1.0 pore volume (i.e. an equivalent drainage of 77 mm).
5.2.1.2 Mean Leachate N Recovery in the Slurry Experiment

The mean recovery of NH$_4^+$-N in the slurry experiment (Figure 4.7) was not as high as the percentage recovery of NO$_3^-$-N in Figure 4.6. This may be due to the NH$_4^+$ ion being adsorbed by cation exchange reactions in the soil. Soil clay and organic matter have a predominantly negative charge and are able to attract and hold positively charged cations such as NH$_4^+$ by the process of cation exchange (Thomas, 1977). However since some ammonium was leached through the soil this would tend to indicate that the cation exchange capacity was insufficient to retain all of the ammonium applied.

The longer period of the slurry experiment (5-32 days) may have allowed nitrogen transformations and losses (e.g. denitrification) to occur which would also result in a lower recovery of N in the leachate.

5.2.2 Soil N Recovery

The highest amount of mean extractable-N was recovered from the top 50 mm of soil (Figure 4.8). A greater amount of ammonium was recovered than nitrate. Low soil NO$_3^-$ levels in the slurry treated columns were assumed to be due to low nitrification rates over the period of column leaching. In a similar study following urine deposition by Holland and During (1978), nitrification rates were reported to be lowered due to the high concentration of ammonium inhibiting the oxidation of ammonium to nitrate by Nitrosomonas spp. Such a mechanism may have also been in operation in the slurry treated soil used in this case study. Although soil nitrite (NO$_2^-$) levels were not measured separately in this investigation, it is also suggested that the high concentration of NH$_4^+$ and free NH$_3$ may also have inhibited the oxidation of any nitrite to nitrate by Nitrobacter spp., in the period immediately after the slurry application.

The very high soil NH$_4^+$ levels measured could have been due to urea hydrolysis increasing the concentration of NH$_4^+$ in soil solution. Quin (1977) reported that a considerable portion of the NH$_4^+$ released from urine-urea hydrolysis in the water-soluble NH$_4^+$-N form. Although water-soluble NH$_4^+$ is rapidly exchanged from the soil solution to soil colloid exchange sites, this form of N is just as susceptible to leaching as anions such as NO$_3^-$ and Cl$^-$, and relatively high levels of NH$_4^+$ would be expected in the leachate collected from the soils receiving excessive NH$_4^+$ applications.

Such high levels of soil NH$_4^+$ represent a considerable potential for further leaching losses of either water-soluble NH$_4^+$ or NO$_3^-$. Ball et al. (1979) found that profile NH$_4^+$ was completely nitrified 21 days after urea was applied to pasture at a rate of 300 and 600 kgN/ha. By 53 days considerable amounts of the applied N had been lost from the 0-450 mm profile.
Thompson and Coup (1943) reported that soil NH$_4^+$ levels which increased from less than 10 $\mu$gN/ml to several hundred $\mu$gN/ml four days after urine application, gradually decreased thereafter until after 14 weeks it reached a level similar to that in control plots.

Such rapid decline in soil NH$_4^+$ levels would greatly increase the potential for N loss via nitrate leaching. Nitrate levels yielded from the nitrification process would most frequently exceed immediate plant requirements and the surplus could quite easily be lost in through-drainage causing groundwater pollution or an economic loss to the farming system.

5.3 Implications for Nitrogen Losses from Land Application

The potential magnitude of N lost after application of slurry can be estimated from the results obtained from the standard solution and the slurry experiments in this case study. Table 5.2 and 5.3 show the estimated mean N losses from the case study site on Lismore stony silt loam soil in the standard solution and the slurry experiments respectively.

The most important variable controlling the potential N losses during winter is that of net rainfall. Calculations assumed rainfall or irrigation occur directly after the slurry application, and do not take plant N uptake into account. Both rainfall amount and continuity need to be considered. Storm events that supply periodic pulses of water to the soil could decrease the leaching efficiency of applied N i.e. small amounts of frequent rainfall would carry much of the applied N from the surface soil layer into the A or even the B horizon. Upon the cessation of rainfall, and so solute movement, diffusive flow of solute N into intra-aggregate pores would lower the amounts of inter-aggregate solute N available for potential leaching. In contrast if continuous rainfall was experienced immediately after slurry application, little of the applied N would be retained in the soil because of the limited size of the diffusive flux.

5.4 The Effect of Local Meteorological Conditions on Leaching

5.4.1 Calculation of Amount of Water Held in the Soil at Field Capacity and Permanent Wilting Point

The average field capacity water content of this soil is 34% (w/w) (Soil Bur. Bull. 26(3)). The average water content at permanent wilting point is 11.5% (w/w) (Kear et al., 1967).
Volumetric water contents ($\Theta_V$) were calculated from these values and the measured bulk density values (Table 4.1) according to the following equation:

$$\Theta_V = \Theta_M \cdot \text{Bulk density} \quad (5.1)$$

Table 5.2 Estimated mean N losses from the case study site following rainfall during winter, or irrigation application, in the standard solution experiment (results calculated using breakthrough N and leachate N recovery curves).

<table>
<thead>
<tr>
<th>Pore volumes of drainage</th>
<th>Equivalent drainage (mm)</th>
<th>Solution Concentration ($\mu gN/ml$)</th>
<th>Amount of N in leachate (kgN/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.1</td>
<td>7.7</td>
<td>58.2</td>
<td>4.46</td>
</tr>
<tr>
<td>0.2</td>
<td>15.4</td>
<td>108.57</td>
<td>8.34</td>
</tr>
<tr>
<td>0.3</td>
<td>23.1</td>
<td>126.2</td>
<td>9.69</td>
</tr>
<tr>
<td>0.4</td>
<td>30.8</td>
<td>120.56</td>
<td>9.26</td>
</tr>
<tr>
<td>0.5</td>
<td>38.5</td>
<td>103.1</td>
<td>7.92</td>
</tr>
<tr>
<td>0.6</td>
<td>46.2</td>
<td>76.66</td>
<td>5.89</td>
</tr>
<tr>
<td>0.7</td>
<td>53.9</td>
<td>56.3</td>
<td>4.33</td>
</tr>
<tr>
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<td>61.6</td>
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<td>3.24</td>
</tr>
<tr>
<td>0.9</td>
<td>69.3</td>
<td>31.27</td>
<td>2.4</td>
</tr>
<tr>
<td>1.0</td>
<td>77.0</td>
<td>25.23</td>
<td>1.94</td>
</tr>
<tr>
<td>1.1</td>
<td>84.7</td>
<td>21.8</td>
<td>1.67</td>
</tr>
<tr>
<td>1.2</td>
<td>92.4</td>
<td>17.94</td>
<td>1.38</td>
</tr>
<tr>
<td>1.3</td>
<td>100.1</td>
<td>13.36</td>
<td>1.03</td>
</tr>
<tr>
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<td>107.8</td>
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<td>0.79</td>
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<td>115.5</td>
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<td>0.68</td>
</tr>
<tr>
<td>1.7</td>
<td>130.9</td>
<td>8.73</td>
<td>0.67</td>
</tr>
<tr>
<td>1.8</td>
<td>138.6</td>
<td>8.53</td>
<td>0.66</td>
</tr>
<tr>
<td>1.9</td>
<td>146.3</td>
<td>8.11</td>
<td>0.62</td>
</tr>
<tr>
<td>2.0</td>
<td>154.0</td>
<td>7.59</td>
<td>0.58</td>
</tr>
<tr>
<td>2.1</td>
<td>161.7</td>
<td>7.53</td>
<td>0.58</td>
</tr>
<tr>
<td>2.2</td>
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<td>6.91</td>
<td>0.53</td>
</tr>
<tr>
<td>2.3</td>
<td>177.1</td>
<td>6.34</td>
<td>0.49</td>
</tr>
<tr>
<td>2.4</td>
<td>184.8</td>
<td>6.85</td>
<td>0.53</td>
</tr>
<tr>
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<td>192.5</td>
<td>6.62</td>
<td>0.51</td>
</tr>
<tr>
<td>2.6</td>
<td>200.2</td>
<td>6.65</td>
<td>0.51</td>
</tr>
<tr>
<td>2.7</td>
<td>207.9</td>
<td>6.17</td>
<td>0.47</td>
</tr>
<tr>
<td>2.8</td>
<td>215.6</td>
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</tr>
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<td>2.9</td>
<td>223.3</td>
<td>5.86</td>
<td>0.45</td>
</tr>
<tr>
<td>3.0</td>
<td>231.0</td>
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<td>0.43</td>
</tr>
<tr>
<td>3.1</td>
<td>238.7</td>
<td>5.5</td>
<td>0.42</td>
</tr>
<tr>
<td>3.2</td>
<td>246.4</td>
<td>5.45</td>
<td>0.42</td>
</tr>
<tr>
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<td>0.36</td>
</tr>
<tr>
<td>3.4</td>
<td>261.8</td>
<td>5.37</td>
<td>0.41</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td></td>
<td></td>
<td><strong>73.0</strong></td>
</tr>
</tbody>
</table>
Table 5.3 Estimated mean N losses from the case study site following rainfall during winter, or irrigation application, in the slurry experiment (results calculated using breakthrough N and leachate N recovery curves).

<table>
<thead>
<tr>
<th>Pore volume</th>
<th>Equivalent drainage (mm)</th>
<th>Solution Concentration (μgN/ml)</th>
<th>Amount of N in leachate (kgN/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.1</td>
<td>8.1</td>
<td>9.48</td>
<td>0.77</td>
</tr>
<tr>
<td>0.2</td>
<td>16.2</td>
<td>9.32</td>
<td>0.76</td>
</tr>
<tr>
<td>0.3</td>
<td>24.3</td>
<td>7.6</td>
<td>0.62</td>
</tr>
<tr>
<td>0.4</td>
<td>32.4</td>
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<td>1.03</td>
</tr>
<tr>
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<td>40.5</td>
<td>22.84</td>
<td>1.85</td>
</tr>
<tr>
<td>0.6</td>
<td>48.6</td>
<td>29.44</td>
<td>2.38</td>
</tr>
<tr>
<td>0.7</td>
<td>56.7</td>
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<td>2.71</td>
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<tr>
<td>0.8</td>
<td>64.8</td>
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<tr>
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<td>72.9</td>
<td>37.84</td>
<td>3.07</td>
</tr>
<tr>
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<td>81.0</td>
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<td>27.24</td>
<td>2.25</td>
</tr>
<tr>
<td>1.3</td>
<td>105.3</td>
<td>27.52</td>
<td>2.23</td>
</tr>
<tr>
<td>1.4</td>
<td>113.4</td>
<td>26.64</td>
<td>2.16</td>
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<td>1.84</td>
</tr>
<tr>
<td>1.6</td>
<td>129.6</td>
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<td>1.57</td>
</tr>
<tr>
<td>1.7</td>
<td>137.7</td>
<td>16.82</td>
<td>1.36</td>
</tr>
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<td>1.8</td>
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<td>1.31</td>
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<tr>
<td>1.9</td>
<td>153.9</td>
<td>16.82</td>
<td>1.36</td>
</tr>
<tr>
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<td>162.0</td>
<td>16.92</td>
<td>1.37</td>
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<td>170.1</td>
<td>16.44</td>
<td>1.33</td>
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<td>2.2</td>
<td>178.2</td>
<td>16.2</td>
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<tr>
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<td>186.3</td>
<td>16.78</td>
<td>1.36</td>
</tr>
<tr>
<td>2.4</td>
<td>194.4</td>
<td>15.64</td>
<td>1.27</td>
</tr>
<tr>
<td>2.5</td>
<td>202.5</td>
<td>14.24</td>
<td>1.15</td>
</tr>
<tr>
<td>2.6</td>
<td>210.6</td>
<td>13.34</td>
<td>1.08</td>
</tr>
<tr>
<td>2.7</td>
<td>218.7</td>
<td>14.55</td>
<td>1.18</td>
</tr>
<tr>
<td>2.8</td>
<td>226.8</td>
<td>14.33</td>
<td>1.16</td>
</tr>
<tr>
<td>2.9</td>
<td>234.9</td>
<td>14.4</td>
<td>1.17</td>
</tr>
<tr>
<td>3.0</td>
<td>243.0</td>
<td>14.1</td>
<td>1.14</td>
</tr>
<tr>
<td>3.1</td>
<td>251.1</td>
<td>11.53</td>
<td>0.93</td>
</tr>
<tr>
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<td>1.02</td>
</tr>
<tr>
<td>3.3</td>
<td>267.3</td>
<td>12.0</td>
<td>0.97</td>
</tr>
<tr>
<td>3.4</td>
<td>275.4</td>
<td>11.5</td>
<td>0.93</td>
</tr>
</tbody>
</table>

Total 53.2

Bulk density values were measured at the site (see Table 4.1) and an average value for the top soil was used for simplicity in these calculations. Table 5.4 gives the volumetric water content values and the amount of water stored in 200 mm soil depth at field capacity and at permanent wilting point.
Table 5.4 Volumetric Water Content ($\Theta_V$) and Amount of Water Stored in the Soil Profile

<table>
<thead>
<tr>
<th>Soil status</th>
<th>$\Theta_M$ (%)</th>
<th>Bulk density g/cm$^3$</th>
<th>$\Theta_V$ (%)</th>
<th>Amount of water (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Field capacity</td>
<td>34</td>
<td>1.6</td>
<td>54.4</td>
<td>109</td>
</tr>
<tr>
<td>Permanent wilting point</td>
<td>11.5</td>
<td>1.6</td>
<td>18.4</td>
<td>37</td>
</tr>
</tbody>
</table>

Note: Values of FC and PWP water contents excluded stones while bulk density may have included some stones hence there could be some error in the calculated values in this table.

Two possible cases will be considered:

**Case 1.** The soil is assumed to be at 'field capacity' at the start of leaching (i.e. any net water input will be lost as drainage).

**Case 2.** The soil is assumed to be at 'permanent wilting point' at the time of slurry application and the start of leaching (i.e. the difference in amount of water stored between field capacity and permanent wilting point has to be added to re-wet the soil profile to field capacity before leaching will occur). To re-wet the soil profile to field capacity before leaching the amount of water required is calculated as:

\[
\text{Amount of water required} = \text{Field Capacity (}$\Theta_V$\text{)} - \text{Permanent Wilting Point (}$\Theta_V$\text{)} \\
= 109 - 37 \text{ (from Table 5.4)} \\
= 72 \text{ mm}
\]

**5.4.2 Leaching if Average Rainfall and Evapotranspiration occurs**

Figure 5.2 shows the average monthly rainfall and ET for 21 years at the case study site. It is evident from the figure that rainfall exceeded evapotranspiration from the month of April to August. Table 5.5 presents the summary of the results obtained after considering the average rainfall and ET for 21 years over the months. It was assumed that the leaching would occur when the rainfall exceeds the ET.
Figure 5.2 Average rainfall and ET for 21 years at Burnham over the months
Table 5.5 Summary of the results if average rainfall and ET occurs at the case study site

<table>
<thead>
<tr>
<th>Case</th>
<th>Solution</th>
<th>Slurry</th>
<th>Rain depth</th>
<th>FC</th>
<th>PWP</th>
<th>Deficit</th>
<th>Drainage</th>
<th>Equivalent pore vol.*</th>
<th>Max. average conc. (µgN/ml)</th>
<th>Relative amount leached (%)</th>
<th>Amount of loss (kgN/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>293</td>
<td>140</td>
<td>153</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>0</td>
<td>188</td>
<td>2.4</td>
<td>c.95</td>
<td>70.0</td>
</tr>
<tr>
<td></td>
<td>293</td>
<td>140</td>
<td>153</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>0</td>
<td>188</td>
<td>2.3</td>
<td>c.6</td>
<td>40.0</td>
</tr>
<tr>
<td></td>
<td>293</td>
<td>140</td>
<td>153</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>72</td>
<td>116</td>
<td>1.5</td>
<td>c.90</td>
<td>65.0</td>
</tr>
<tr>
<td></td>
<td>293</td>
<td>140</td>
<td>153</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>72</td>
<td>116</td>
<td>1.4</td>
<td>c.4</td>
<td>38.0</td>
</tr>
</tbody>
</table>

* 1 pore volume in the solution experiment = 195.4
* 1 pore volume in the slurry experiment = 204.6
In case 1, rainfall between April to August will result in a leaching equivalent to 2.4 pore volumes in the standard solution experiment. Figure 4.1 shows that in the standard solution experiment the peak concentration of nitrate has already been leached by then. In other words, under conditions of average rainfall and average evapotranspiration the entire nitrate addition would leach through the soil (Figure 4.6).

In the slurry experiment (Case 1) the net water input from April to August will result in a leaching equivalent to 2.3 pore volumes. Figure 4.2 shows that in the slurry experiment the peak ammonium concentration had already leached through the soil by then.

In case 2, the rainfall between April and August will represent 1.5 pore volumes in the standard solution experiment by which time most (>90%) of the nitrate had leached through (Figure 4.6). In the slurry experiment the same amount of water will represent 1.4 pore volumes of drainage. By this time the peak concentration had already leached through the soil.

5.4.3 Leaching if Maximum Rainfall and Evapotranspiration occurs

Figure 5.3 shows the maximum monthly rainfall and ET for 21 years at the case study site. Rainfall exceeds evapotranspiration from March to September. Table 5.6 shows the summary of the results obtained after considering the maximum rainfall and ET for 21 years.

In case 1, the net amount of water input will represent 7.9 pore volumes in the standard solution experiment. Since all of the nitrate is leached by 2.7 pore volumes then this excess water will move it further through the underlying gravels. In the slurry experiment the net water input will represent 7.5 pore volumes of drainage. Figure 4.2 shows that ammonium is still being leached after 3.3 pore volumes, however it is unknown whether the addition of 7.5 pore volumes will continue the leaching. Considering the amount of NH$_4^+$-N recovered from the soil (Figure 4.8) this does however appear likely.

In case 2, the net water input in the standard solution experiment and slurry experiment represent 7.0 and 6.6 pore volumes, respectively. As discussed in Case 1, the leaching of nitrate from the soil will be completed before this time and the excess water will simply move the nitrogen further through the underlying gravels.
Figure 5.3 Maximum rainfall and ET for 21 years at Burnham over the months.
Table 5.6 Summary of the results if maximum rainfall and ET occurs at the case study site

<table>
<thead>
<tr>
<th></th>
<th>Rain</th>
<th>ET</th>
<th>Net Rain</th>
<th>Slurry depth</th>
<th>Soil water content</th>
<th>Deficit</th>
<th>Drainage</th>
<th>Equivalent pore vol.*</th>
<th>Max. average conc.</th>
<th>Relative amount leached</th>
<th>Amount of loss</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(mm)</td>
<td>(mm)</td>
<td>(mm)</td>
<td>(mm)</td>
<td>(mm)</td>
<td>(mm)</td>
<td>(mm)</td>
<td>(μgN/ml) (%)</td>
<td>(kgN/ha)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Case 1</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Solution</td>
<td>1007</td>
<td>429</td>
<td>578</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>0</td>
<td>613</td>
<td>7.9</td>
<td>126.2</td>
<td>100  73.0</td>
</tr>
<tr>
<td>Slurry</td>
<td>1007</td>
<td>429</td>
<td>578</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>0</td>
<td>613</td>
<td>7.5</td>
<td>37.94</td>
<td>&gt;8.0 &gt;55.8</td>
</tr>
<tr>
<td><strong>Case 2</strong></td>
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<td></td>
<td></td>
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</tr>
<tr>
<td>Solution</td>
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<td>429</td>
<td>578</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>72</td>
<td>541</td>
<td>7.0</td>
<td>126.2</td>
<td>100  73.0</td>
</tr>
<tr>
<td>Slurry</td>
<td>1007</td>
<td>429</td>
<td>578</td>
<td>35</td>
<td>109</td>
<td>37</td>
<td>72</td>
<td>541</td>
<td>6.6</td>
<td>37.94</td>
<td>&gt;8.0 &gt;55.8</td>
</tr>
</tbody>
</table>

* 1 pore volume in the solution experiment = 195.4
* 1 pore volume in the slurry experiment = 204.6
5.4.4 Leaching if Minimum Rainfall and Evapotranspiration occurs

Figure 5.4 shows the minimum monthly rainfall and ET for 21 years at the case study site. Under these circumstances rainfall will exceed the evapotranspiration in the months of July and October only. Table 5.7 shows the summary of the results obtained after considering this case. The leaching is supposed to have occurred when the rainfall exceeded the ET.

In Case 1, the net amount of water input will represent 0.7 and 0.6 pore volume in the standard solution and slurry experiments respectively. Figure 4.1 shows that by 0.7 pore volumes the peak nitrate concentration had occurred in the leachate, however the nitrate concentration remained high until approximately 1.5 pore volume. Therefore, under the small rainfall input a large amount of nitrate will remain in the soil. Figure 4.2 shows that by 0.6 pore volume a large amount of ammonium had not been leached and that the peak concentration had also not appeared in the leachate. A large amount of ammonium would therefore be left in the soil.

In Case 2 when the soil is assumed to be at permanent wilting point the amount of net input is only 52 mm, whereas 72 mm of input is required to bring the soil to field capacity. As a result only a minimal amount of leaching/drainage is likely to occur. The entire amount of nitrate and ammonium will be left in the soil.

5.5 Further Considerations

Francis (1986) using a Cl tracer found that the leaching efficiency was lower if the solute had been allowed time to diffuse into the surface soil aggregates for 48 hours prior to water application. Thus if rainfall or irrigation occurred some time (e.g. 48 hours) after slurry application then the NO$_3^-$-N losses would be lower.

Losses may also be less if soils are drier than 'field Capacity' prior to the irrigation or rainfall event. Macropore flow may however occur with rainfall irrigation intensities as low as 1-10 mm/hr (Beven and Germann, 1982) hence some losses may still occur.

The amounts of NO$_3^-$ leached typically increases as the rate of waste application increases (Bielby et al., 1973; Sherwood, 1981b). Sherwood (1981b) applied pig slurry to grassland over a four year period at rates of nil, 400, 700, and 1400 kgN/ha/yr and estimated leaching losses as 0.9, 1.8, 77, and 162 kgN/ha/yr respectively. In some cases the percentage of applied N lost in leachate increases with increasing application rates (Spallacci, 1981) while in others it decreases (Dam Kofoed, 1979).
Figure 5.4 Minimum rainfall and ET for 21 years at Burnham over the months
Table 5.7 Summary of the results if minimum rainfall and ET occurs at the case study site

<table>
<thead>
<tr>
<th></th>
<th>Rain</th>
<th>ET</th>
<th>Net Rain</th>
<th>Slurry depth</th>
<th>Soil water content</th>
<th>Deficit</th>
<th>Drainage</th>
<th>Equivalent pore vol.*</th>
<th>Max. average conc. (μgN/ml)</th>
<th>Relative amount leached (%)</th>
<th>Amount of loss (kgN/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Case 1</strong></td>
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<td></td>
<td></td>
</tr>
<tr>
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<td>109</td>
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<td>126.2</td>
<td>c.70</td>
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<td>29.44</td>
<td>c.10</td>
<td>c.70</td>
</tr>
</tbody>
</table>

* 1 pore volume in the solution experiment = 195.4
* 1 pore volume in the slurry experiment = 204.6
Field et al. (1985a) reported that most of the nitrate accumulated from spring fertiliser application and summer mineralisation was leached below the plant uptake zone in late autumn and winter as a result of rainfall inputs during these periods. Field et al. (1985b) further concluded that nitrate, equivalent to about 20% of that applied in urine, was leached out of 1 m deep soil columns during the following winter and spring. By the end of next drainage season the equivalent of 48% of all N applied had been collected in the drainage water. Cameron et al. (1989) reported that the leaching losses from applied nitrate are slightly greater when continuous rather than intermittent rainfall occurs.

5.6 Suggestions for Future Work

While conducting this case study the following requirements were felt to be necessary for further work examining leaching losses of nitrogen from pig slurry application to land:

1. The rate and extent of solute leaching needs to be studied under conditions which simulate more closely the natural conditions occurring in the field. This could be achieved by the intermittent application of either simulated rainfall, sprinkler irrigation, or border dyke irrigation.

2. The effects of plant uptake on leaching losses of N in field soils needs to be examined more closely. Undisturbed soil lysimeters with actively growing vegetation present could be used for this purpose.

3. The evaluation of N leaching using a number of existing solute leaching models needs to be undertaken under a variety of input conditions to determine which is the most likely to simulate solute leaching in the field.
CHAPTER 6
CONCLUSIONS

From the experiments and their analysis the following conclusions can be made:

1. Extensive preferential flow through macropores occurred in the standard solution experiment during leaching. A very high mean concentration of approximately 58 μgN/ml was recorded immediately after drainage had started (0.1 pore volume). This shows that macropore movement can result in a high concentration of nitrate being leached from the soil after only a small amount of net water input and drainage. Even if only 10% of the nitrogen present in the slurry is converted into nitrate, as it could be, then the nitrate level of water moving into aquifer would be in excess of the World Health Organisation limits of 10 μgN/ml (WHO, 1971) for drinking water.

2. Following application and leaching of slurry the peak ammonium concentration of the leachate was approximately 35 μgN/ml and occurred at an equivalent of 0.9 pore volume of drainage. Less extensive macropore flow was observed and this was attributed to slurry blocking the soil pores.
REFERENCES


Appendix A
Profile Description of the Lismore Stony Silt Loam

A 0-150mm Very dark greyish brown silt loam with stones; friable; moderately developed fine nutty and granular structure; many roots; indistinct boundary

B1 150-250mm dark yellowish brown; stony silt loam; friable; weakly developed medium nutty structure with many casts; few roots; indistinct boundary

B2 250-350mm olive brown; sandy gravels; compact structureless; many diffuse yellowish brown iron staining among gravels; few roots; diffuse boundary

C >350mm on firm greywacke gravels.

¹Source: Kear et al., (1967) New Zealand Soil Bureau Bulletin 14
### Appendix B

**Rainfall in mm, Burnham Sewage Plant**

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Appendix D

Individual core concentration of NO$_3^-$-N in the standard solution experiment

![Graph of individual core concentration of NO$_3^-$-N](image)

- Core 1 SS

![Graph of individual core concentration of NO$_3^-$-N](image)

- Core 2 SS
Appendix D (contd.)
Appendix E

Individual core concentration of NH$_4^+$-N in the slurry experiment
Appendix E (contd.)

---

**Missing data between 0.4 and 1.2 pore volume**

---

**Core 3 Slurry**

---

**Core 4 Slurry**
Appendix E (contd.)

![Graph showing concentration vs pore volume for Core 5 Slurry](image-url)
Appendix F

Individual core relative concentration of $\text{NO}_3^-$-N in the standard solution experiment.
Appendix F (contd.)

\[
\begin{align*}
\text{Relative Concentration} & \\
\text{Pore volume} & \\
0.1 & 0.3 & 0.5 & 0.7 & 0.9 & 1.1 & 1.3 & 1.5 & 1.7 & 1.9 & 2.1 & 2.3 & 2.5 & 2.7 & 2.9 & 3.1 & 3.3
\end{align*}
\]

--- Core 3 SS

\[
\begin{align*}
\text{Relative Concentration} & \\
\text{Pore volume} & \\
0.1 & 0.3 & 0.5 & 0.7 & 0.9 & 1.1 & 1.3 & 1.5 & 1.7 & 1.9 & 2.1 & 2.3 & 2.5 & 2.7 & 2.9 & 3.1 & 3.3
\end{align*}
\]

--- Core 4 SS
Appendix F (contd.)

![Graph showing relative concentration over pore volume. The graph is labeled as Core 7 SS.](image-url)
Appendix G

Individual core relative concentration of \( \text{NH}_4^+ \) in the slurry experiment

![Graph showing the relative concentration of \( \text{NH}_4^+ \) for Core 1 and Core 2 slurry experiments.](image-url)
Appendix G (contd.)

Missing data between 0.4 and 1.2 pore volume

---

Core 3 Slurry

---

Core 4 Slurry
Appendix H

Individual core native extractable-N in the soil profile

Native extractable-N, Core 1

Native extractable-N, Core 2
Appendix I

Individual core extractable-N in the slurry experiment

- Extractable ammonium
- Extractable nitrate

Extractable-N, Slurry, Core 1

Extractable-N, Slurry, Core 2
Appendix I (contd.)

Extractable-N, Slurry, Core 3

- Extractable ammonium
- Extractable nitrate

Extractable-N, Slurry, Core 4

- Extractable ammonium
- Extractable nitrate
Appendix I (contd.)

![Graph showing extractable N, Slurry, Core 5 at depths of 0-5, 5-10, and 10-15 cm. The graph indicates the distribution of extractable ammonium and nitrate across these layers.](image-url)