THE FATE OF POTASSIUM IN GRAZED DAIRY PASTURES

A thesis presented in partial fulfilment of
the requirements for the degree of
Doctor of Philosophy in Soil Science
at Massey University

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1988

ABSTRACT

Losses of potassium (K) are inevitable in grazed New Zealand dairy pasture. Losses result from the removal of K in animal products, transfer of excreta to the dairy shed and raceways, and K leaching below the effective plant rooting zone. In this study the fate of K in a grazed dairy pasture was investigated in order to quantify the extent of these losses more precisely. As a consequence an improved methodology for predicting K fertiliser requirements has been developed.

A field experiment on the Tokomaru silt loam soil (yellow-grey earth) showed that immediately after a urination event in July up to 59% of the urine K moved preferentially through a network of soil macropores to below a soil depth of 15 cm. The remaining 41% of the urine K was retained in this depth of soil in the form of exchangeable soil K^+. Over the following spring and summer, plants were able to recover the urine K retained in the 0-15 cm depth of soil. The field experiment also showed that pasture plant roots growing in urine affected soil derived their K requirements mainly from the 0-15 cm depth of soil. Any K that moved beyond a depth of 15 cm due to leaching can be considered to be a loss of K from the grazed dairy pasture.

In a laboratory study, two soil types of contrasting mineralogy (the Tokomaru silt loam, a yellow-grey earth, and the Egmont brown loam, a yellow-brown loam) were incubated with either dairy cow urine or potassium chloride. Within 24 hours the soil pH in the urine treatment increased by one unit on both soil types. This increased soil pH was evident for 41 days in the Tokomaru soil and for the duration of the experiment (106 days) in the Egmont soil. For the Egmont soil, which contains larger amounts of pH dependent negative surface charge, this increase in soil pH resulted in greater K^+ adsorption by the soil from the urine treatment compared with the potassium chloride treatment. The Tokomaru soil contained smaller amounts of pH dependent charge, and there was no difference in K^+ adsorption between the two K sources. Such a result suggested that where potassium chloride solutions are used to simulate urine K additions to the soil, misleading results may occur. In the field, K^+ adsorption, and therefore leaching losses due to rainfall, from an application of urine may vary between soil types that differ in their surface charge characteristics.

The redistribution of urine K due to leaching, plant uptake and soil adsorption after an application of dairy cow urine to soil was investigated in two experiments with intact cores of four different soil types. In climatic conditions that simulated those of August/September in the

Manawatu region up to 80% of the urine moved preferentially beyond the 15 cm depth of soil. This preferential movement of urine occurred too quickly for adsorption reactions to occur between the soil surface and the solutes in the urine, therefore the soil type had little effect on the amount of urine K that was leached. After the urine application, sufficient simulated rain (as 6 events of 5 mm per day applied every second day over a 6-8 week period) was applied to produce 300 mm of drainage. Under free draining conditions the rainfall tended to move preferentially through the macropores of the soil cores bypassing the urine K in the soil, and so the amount leached was equivalent to 3-6% of the urine K applied. Where drainage was impeded, the rainfall moved less preferentially and more K (equivalent to 7-27% of the urine K applied) from the soil micropores was leached.

A technique was developed to measure the amount of urine K that was lost preferentially beyond the 15 cm depth of soil in the field situation. This technique was used on seven soil types, four of which were sampled in two different seasons. Results showed that between 0 and 46% of the urine K (and nitrogen) moved beyond the 15 cm depth due to preferential flow. These field results confirmed that preferential flow of urine through soil macropores resulted in the largest loss of K from all the soils studied. In general, the amount of K that was lost was more dependent on the physical conditions of the soil surface (e.g., water repellent or compacted soil surface, irregular surface microtopography) than the soil type or moisture content.

Data generated from this study were used to construct a model (the K Loss model) to predict the extent of K losses from three seasonal supply dairy farms. For one of the farms, on a peat soil, it was possible to show that the predictions of soil K losses made by the K Loss model were comparable with actual changes in soil K status. Similar evaluations on the other sites were impossible because the changes of soil K with time were insignificant in comparison with the native soil K contents.

Comparisons were made between the losses of K estimated by the K Loss model and the model used by the New Zealand Ministry of Agriculture and Fisheries in their Computerised Fertiliser Advisory Service (CFAS). The CFAS model did not account directly for the loss that occurs due to preferential flow of urine beyond the 15 cm depth of soil, although in some cases this loss can be considerable (e.g., 48 kg K ha^{-1} yr^{-1} on the Massey University No. 4 dairy farm). In addition the CFAS model appeared to overestimate the amount of K lost due to transfer of excreta to unproductive areas of the farm (e.g., the transfer loss was overestimated by 39 kg K ha^{-1} on the No. 4 dairy farm). More accurate predictions of the amounts of K lost, and therefore the amounts of K fertiliser required to replace these losses, can be made easily using information supplied by the farmer from which the chronometric detail of the cows excretion pattern can be established.

ACKNOWLEDGEMENTS

I would like to express my gratitude to the following people for their contribution towards this thesis:

Dr Paul Gregg and Dr Mike Hedley for their supervision, guidance and patience during all stages of this project.

Past and present members of the Soil Science Department, particularly Dr Bolan, Dr Dave Scotter and Mr Russ Tillman, for advice, assistance and friendship throughout.

Mr Martin Eggels for technical assistance at different stages of the project.

The people who provided sites for the field trials and data for this study: staff of the Massey University No. 1, 3 and 4 dairy farms, staff of the Ministry of Agriculture and Fisheries Taranaki Agricultural Research Station, Mr Alan Gregg, Mr Paul Phippen, Mr Stuart Hamilton, Mr Ron Falconer, and the farmers who completed the questionnaire.

Dr Jacqueline Rowarth for her friendship and work in proof-reading and photocopying this thesis.

The Ministry of Agriculture and Fisheries for providing study leave which enabled this project to be carried out.

Finally, Lloyd Williams for his patience, support and encouragement throughout.

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CHAPTER 1

INTRODUCTION

Potassium (K) is an essential element for plant growth. Therefore where the supply of K from the soil is likely to limit plant growth, applications of fertiliser K are required.

Topdressing of New Zealand pastures began in the 1880's. It was not until the 1930's, however, that there was any evidence that pasture growth was limited by a K deficiency (Smallfield, 1935). Since then it has been observed that K deficiencies develop with time under continuous dairy farming. The reasons for this have been attributed to the removal of K in animal products (milk and culled animals), hay and silage crops, losses through transfer of excreta around the farm, and leaching (New Zealand Department of Agriculture, 1957).

Associated with this increased incidence of K deficiency has been an increase in the amount of K fertiliser imported annually into the country from 7,400 tonness in 1925 (worth $130,000; Aston, 1926) to an average of 155,000 tonnes in 1985 and 1986 (worth $33 million; Ministry of Agriculture and Fisheries, 1987).

Importation of potassium fertilisers is expensive both to the dairy farmer and to the nation, therefore it is necessary to ensure that K fertiliser is efficiently used. The amount of K fertiliser required to maintain productivity at a desirable level can be predicted by calculating inputs of K to, and losses of K from, the farming system under consideration. To do this it is necessary to describe quantitatively the forms of K found in a grazed dairy pasture ecosystem and understand how K is transferred (cycled) between these forms. In general, K moves from the soil to pasture plants and then back to the soil, either through the death of plant tissue or via the excreta of grazing dairy cows. Together these pathways are referred to as parts of a "cycle" in an agricultural system, although this cycle is only a minor component of the biogeochemical K cycle that occurs on a global scale.

The concept of nutrient cycling in a grazed pasture has been studied over the last 30 years since Sears and his co-workers demonstrated the importance of the grazing animal in returning nutrients ingested in the herbage to the soil in the form of dung and urine (e.g., Sears et al., 1948). While this early research indicated that grazing animals had a beneficial role in nutrient cycling through the transfer of fertility in the form of excreta around the farm (Sears, 1950), this viewpoint has since been modified. More recent research has shown that the high return of nitrogen to the soil in the form of extremely concentrated urine spots can lead to greater losses than originally indicated, particularly on intensively managed, high fertility farms (Ball and Keeney, 1981). The observation that K deficiencies develop under dairy farming (New Zealand Department of Agriculture, 1957) adds support to extend the latter theory to K. Obviously the role of the dairy cow in the K cycle needs to be clarified.

The main objective of this study was to investigate the effects of the grazing animal, soil physical and chemical properties and the plant processes on K cycling, and to integrate these aspects in to a model to explain the fate of K in a grazed dairy pasture.

Most of the previous research on nutrient cycling in New Zealand and other countries has been concerned with nitrogen and phosphorus; few studies have been carried out on the cycling of K in grazed pasture. In New Zealand, a simple model of the K cycle has been developed by Ministry of Agriculture and Fisheries (MAF) staff (Cornforth and Sinclair, 1984). Using a mass balance approach this model estimates the inputs and losses of K that occur for an individual farm. The difference between inputs and losses is used as an estimate of fertiliser requirements in the MAF's Computerised Fertiliser Advisory Service (CFAS). The CFAS model indicates that the greatest loss of K from a dairy farm occurs through the transfer of excretal K to unproductive areas of the farm such as the dairy shed and raceways. In comparison, smaller amounts of K are lost via animal products and leaching. For example, on a typical dairy farm in the Manawatu region with a stocking rate of 3 cows ha^{-1}, the model predicts losses of 69 kg K ha^{-1} yr^{-1} due to transfer to unproductive areas, 15 kg K ha^{-1} yr^{-1} due to removal in animal products and 3 kg K ha^{-1} yr^{-1} due to leaching. To date few attempts have been made to test the accuracy of these predicted losses in the field situation despite the widespread use of the CFAS model in New Zealand by advisers for predicting fertiliser requirements. It is clear that field information on the fate of K in grazed dairy pasture is necessary in order to be able to improve our understanding of the K cycle, to test the accuracy of the CFAS model, and ultimately, to give rational and economic advice to the farmer.

The general objectives of this study were to:

1. survey the current literature describing the forms of potassium and their modes of redistribution within grazed dairy systems.

2. identify areas where further research could improve the understanding of the fate of soil and fertiliser potassium in grazed dairy systems.

3. research and quantify some of the areas identified in (2).

4. use the data obtained in (3) to model the potassium cycle in grazed dairy systems in order to predict the losses of potassium.

5. compare the predicted and actual losses of potassium from case studies with those losses estimated by MAF's CFAS model and to suggest improvements that could be made to the CFAS model.

CHAPTER 2

REVIEW OF LITERATURE

2.1 INTRODUCTION

The form and function of the K cycle in a grazed dairy pasture is strongly influenced by soil minerals, soil water, climate, plants, animals and farm management. It is necessary to understand the interrelationship of all these factors before it is possible to describe the fate of fertiliser K on dairy farms. Such a description is the first step towards predicting fertiliser requirements.

Potassium fertiliser is applied to dairy pasture to promote plant growth (dry matter production) rather than to provide a source of K for animal health. Fertilising with K will stimulate increased nitrogen (N) fixation by legumes (Lanyon and Smith, 1985) and improve nitrate (NO_3^-) uptake by grass species (Mengel and Kirkby, 1987). The amount of K fertiliser required varies between individual farms according to the above factors which influence the K cycle. A simplified diagram showing the main compartments and flows of K in a grazed pasture is presented in Figure 2.1. In terms of the mechanisms of gains and losses, the K cycle can be summarised simply as follows. Gains to the cycle usually enter through the soil compartment via the application of fertiliser, and the deposition of K in rainfall and irrigation water. The plant available portion of the soil compartment is that K^+ which is present in soil solution or is adsorbed on soil surfaces and is exchangeable with soil solution K^+. Plant available K can be derived from K gains to the cycle, from release of K during decomposition of plant and animal residues, and soil minerals. The soil compartment is also a medium for losses of K due to leaching.

The amount of K in the plant compartment is a function of the ability of the plant to absorb soil solution K^+ and the size of the exchangeable K^+ pool in the soil. Plant K can be consumed by the grazing animal, lost from the cycle through removal in hay and silage crops, or returned to the soil through plant and litter decomposition.

The grazing animals exert a major influence on the spatial redistribution of K through their variable patterns of herbage ingestion and excreta return. Losses of K from the animal compartment occur due to removal of animal products from the farm and transfer of excreta to areas outside the cycle such as in raceways and the dairy shed.

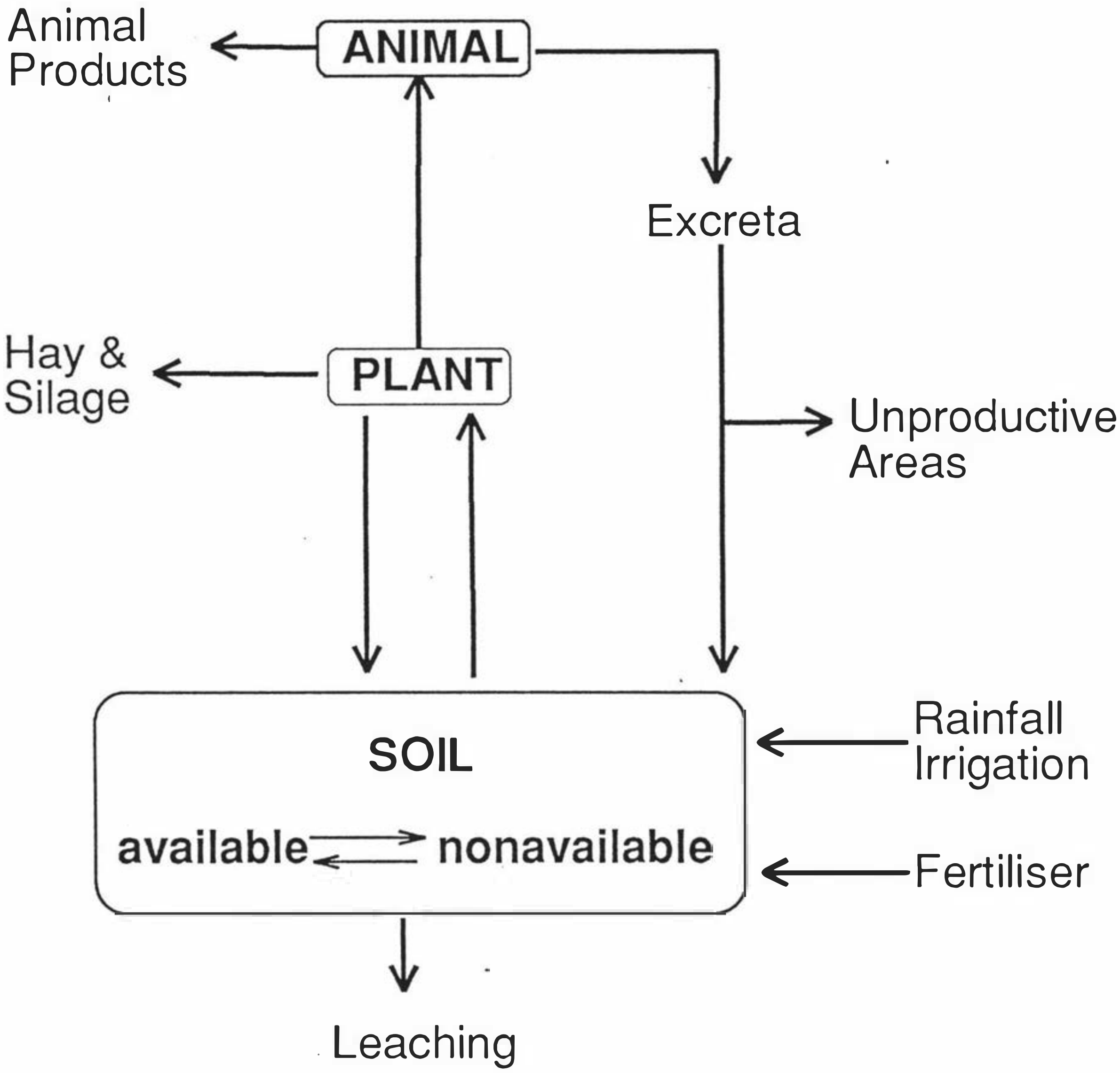


Figure 2.1 Compartments and transformations of the K cycle in a fertilised, grazed pasture.

This review will quantify and define the three compartments of the cycle (sections 2.2-2.4). Next the gains of K that enter the cycle (section 2.5), the transformations that take place between these compartments (section 2.6) and the losses that occur from the cycle (section 2.7) will be considered. The information on the quantity and rate of transfer of K along these pathways can be used to develop a model of the K cycle which can be used to predict fertiliser recommendations. The final part of the literature review (section 2.8) will discuss the attempts that have been made to model the K cycle.

2.2 SOIL POTASSIUM

The soil compartment is an important part of the K cycle as this compartment affects the supply of K to the plants and the amount of K that is leached from the cycle. The latter is a major pathway for loss from the cycle.

Soil K has been the object of many research projects and several detailed reviews have been written on this subject (e.g., Reitemeier, 1951; Arnold, 1960, 1962; Rich, 1968; Metson, 1980; Lee and Metson, 1981; Sparks and Huang, 1985; Sparks, 1987). This review will briefly describe the forms of K that occur in the soil, their importance in the K cycle as sources of plant K, and the transformations that occur from one form of soil K to another.

Potassium exists in soils in inorganic forms only, the majority of which is not available to plants. Four forms of soil K are recognised. In order of increasing availability for plant uptake these forms are structural, nonexchangeable, exchangeable and water soluble (Sparks, 1987). The latter three forms of K are interrelated (Figure 2.2) so that a change in the amount of one form is compensated for by changes in one or all of the other forms (Reitemeier, 1951).

2.2.1 Structural Potassium

The majority of K in the soil is present in the structural form (Metson, 1980). In other publications structural K is also referred to as mineral, inert or unavailable K. Structural K occurs in the primary soil minerals mica, feldspar and volcanic glass, which are derived from soil parent materials (Metson, 1980).

Micas are layered silicate materials made up of layers which comprise an octahedral hydroxide sheet sandwiched between two tetrahedral oxide sheets (often referred to as a 2:1 structure) (Fanning and Keramidas, 1977). Isomorphic substitution of silicon by aluminium in

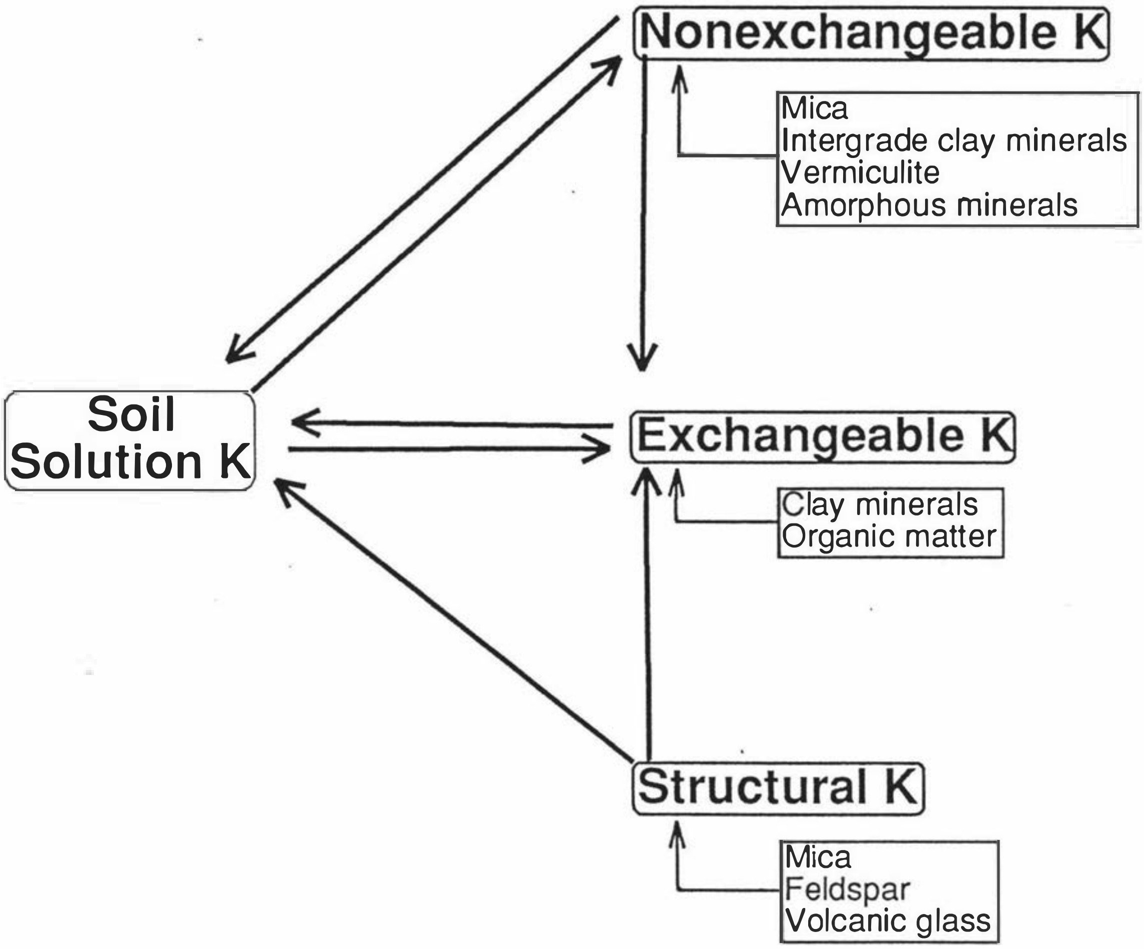


Figure 2.2 Forms of soil K and their interrelationships (Arnold, 1962; Sparks, 1987).

the tetrahedral layers results in negative charges which are balanced by cations such as K^+, sodium (Na^+) and calcium (Ca^{++}) (Fieldes and Swindale, 1954). The two micas which contain K^+ in their interlayers are muscovite and biotite, and these have K contents of 80 and 70 g K kg^{-1}, respectively (Sparks, 1987). In New Zealand soils, micaceous minerals are found mainly in the clay fraction of brown-grey earths, yellow-grey earths and the lesser weathered yellow-brown earths (Fieldes and Swindale, 1954).

Feldspars are constructed of three dimensional honeycomb frameworks of linked silica and alumina tetrahedra, with cations situated in the intervening spaces amongst the framework (Sparks, 1987). The feldspars, which contain K^+ as the main cation (110 g K kg^{-1}) are orthoclase and microcline (Sparks, 1987), and are found in many New Zealand soils particularly in the coarser soil particles (e.g., silt and sand; Fieldes and Swindale, 1954).

Volcanic glass is found in yellow-brown loam (basic volcanic glass) and yellow-brown pumice (acidic volcanic glass) soils in New Zealand (Fieldes and Swindale, 1954). The structure of volcanic glass comprises a random network of silica tetrahedra and cations (Fieldes and Swindale, 1954).

With time, the primary minerals weather to form secondary minerals which are mainly found in the clay fraction of the soil. The degree of weathering depends not only on the environment but also the type of primary mineral. Biotite and basic volcanic glass are the most easily weathered, while the feldspars and acidic volcanic glass are only slowly weathered (Fieldes and Swindale, 1954). The products of the weathering process are amorphous hydroxides from feldspars and volcanic glass, and hydrous micas (e.g., illite, vermiculite and smectite) from the micas (Fieldes and Swindale, 1954).

During the weathering of micaceous minerals, K^+ is replaced by hydrated cations either layer by layer (layer weathering) or from the edges of several layers at once (edge weathering; Fanning and Keramidas, 1977). As weathering progresses there is a decrease in K^+ content, and an increase in water content, specific surface area and cation exchange capacity (CEC) of the minerals (Fieldes and Swindale, 1954). Thus while unweathered micas are sources of K^+ they have little ability to adsorb K^+ from the soil solution due to their low CEC. Intermediate weathering products such as illite are both a source of K^+ and a sink for adsorbed K^+, while the expanded clay minerals vermiculite and smectite that are the end products of weathering, are sinks and not sources of K^+. Other layered silicates which are not of the 2:1 structure but which are also end products of weathering micas, such as chlorite, are intermediate in CEC (similar to illite; Barnhisel, 1977). Kaolinite and halloysite (end products of weathering processes under acid conditions) have very low CEC and therefore have only a poor affinity for K^+ adsorption (Dixon, 1977).

A considerable variation exists in the type of minerals which are present in New Zealand soils due to differences in the parent material and the degree of weathering (Metson, 1980). Such a variation has resulted in a range of K contents in the top 30 cm of the soil profile from 2.5 to 55 tonnes ha^{-1} (During, 1984). Under most conditions the availability of structural K for plant growth is very slow (Selim et al., 1976).

2.2.2 Nonexchangeable Potassium

Nonexchangeable K^+ is held between layers of micaceous clays in positions not readily accessible for exchange with solution cations. Such positions can be either between structural layers of micas, intergrade hydrous micas (e.g., illite) and vermiculites, or in the wedge zones during edge weathering of micas (Figure 2.3). Due to the position and affinity for K^+ by the clay surfaces other cations do not exchange easily with the K^+ (Sparks, 1987) and so the K^+ is referred to as nonexchangeable, difficultly exchangeable or reserve K^+. Fully expanded clay minerals such as smectite, kaolinite and halloysite do not contain this form of K^+ (Thomas and Hipp, 1968).

Nonexchangeable K^+ also includes K^+ held by amorphous hydrous aluminosilicates such as allophane (Sparks, 1987). Allophane naturally has a low K content (1 g K kg^{-1}) but is recognised as having a high affinity for retaining K^+ in a nonexchangeable form through retention within its structural configuration (Sparks, 1987).

Nonexchangeable K^+ is capable of slowly becoming plant available particularly if solution K^+ levels are decreased (Doll and Lucas, 1973; Barber, 1984; Robinson, 1985). Measurements have shown that nonexchangeable K^+ can be released into solution when the concentration of K^+ in the soil solution is less than 2 μmole dm^{-3} (Barber, 1985) and <5 μg ml^{-1} (Jackson and During, 1979).

The role of nonexchangeable K^+ as a source of plant available K^+ has been demonstrated in many experiments where the measured losses of K^+ through plant uptake and leaching were greater than the measured changes in exchangeable K^+ plus fertiliser inputs (Hay et al., 1976; Robinson, 1985). In experiments on New Zealand soils containing micaceous minerals the rate of release of nonexchangeable K^+ has been predicted at being

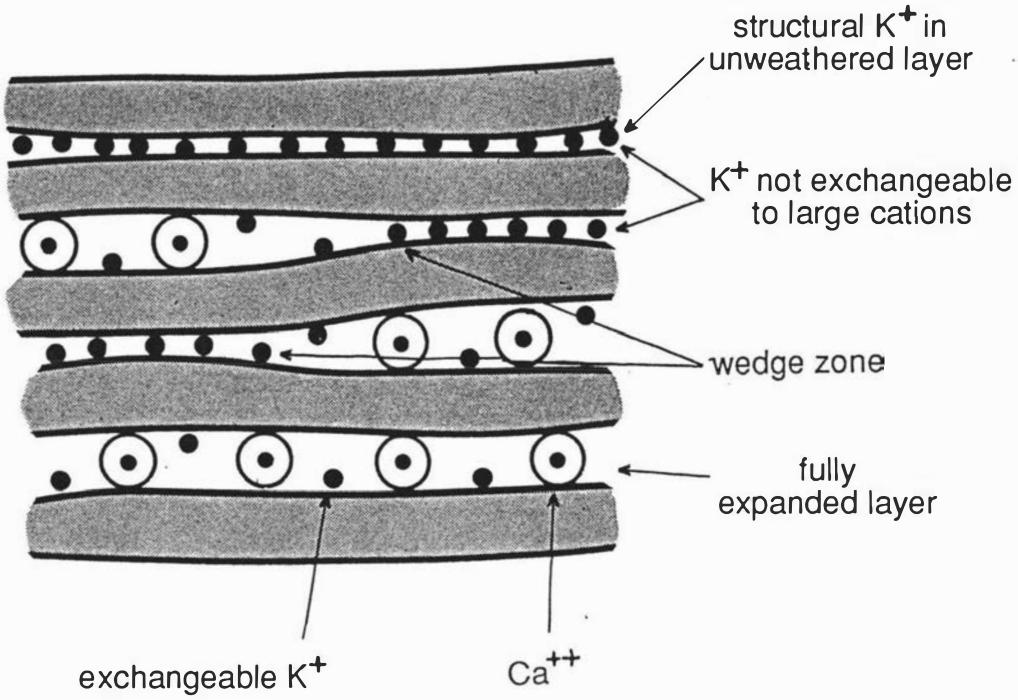


Figure 2.3 Schematic diagram of the positions of K found in micaceous minerals (Rich, 1968).

between 100 and 200 kg K ha^{-1} yr^{-1} (Metson and Hurst, 1953; Lee and Metson, 1977; Weeda, 1979; During and Campkin, 1980). The rate of release of nonexchangeable K$^+$ depends on the rate of removal of the K$^+$ through plant uptake and leaching, as well as on the type of minerals present. For example, while the rate of release of K$^+$ from the weathering of biotite has been shown to be sufficient to meet plant requirements, the rate of release of K$^+$ from muscovite was slower and supplementary fertiliser K was required (McLean and Watson, 1985).

2.2.3 Exchangeable Potassium

Exchangeable K$^+$ is electrostatically bonded by the surface negative charges on clays and organic matter. These charges are referred to as cation exchange sites and arise in several ways. On micaceous clays they occur through the substitution of one structural cation for another, as described in section 2.2.1. The degree of substitution and therefore the CEC varies between mineral species (Thomas and Hipp, 1968). Additional negative charges are created by the presence of weak acids on clay surfaces (Thomas and Hipp, 1968). While weak acid sites can be found in crystalline clays they are more common in amorphous clay minerals (e.g., allophane) where they can make up the majority of the clay's negative charge sites (Thomas and Hipp, 1968). The negative charge of organic matter is derived from carboxyl and phenolic groups which behave like weak acids (Thomas and Hipp, 1968).

The amount of exchangeable K$^+$ present in a soil will, in part, depend upon the quantity of soil surface negative charge. Whereas negative charge arising from isomorphous substitution in clay minerals remains constant, the negative charge arising from weak acid groups varies with soil pH (pH dependent charge)(Thomas and Hipp, 1968). The amount of negative charge on the organic matter, on the edge surfaces of the amorphous clay minerals and to a lesser extent the crystalline clay minerals, increases as pH increases due to dissociation of protons from weak acid groups (Gast, 1977). In some New Zealand soils which contain large amounts of pH dependent negative charge (e.g., yellow-brown loams), increasing the soil pH by one unit may raise the CEC by 15 me 100 g^{-1} (Edmeades, 1982).

The affinity of the exchange sites for K$^+$ is related to the nature of the soil surface (highest on 2:1 clay minerals and lowest on organic matter) and the concentration of K$^+$ relative to other exchangeable ions present in the soil especially the divalent cations (Barber, 1984). Where many of the exchange sites are filled with Ca^{++} and magnesium (Mg^{++}), K$^+$ moves to the more weakly bonded sites (Barber, 1984).

In the major dairy farming soils of New Zealand exchangeable K^+ levels account for between 1 and 5% of the total soil K content (Table 2.1). Exchangeable K^+ exchanges easily with other cations from soil solution and so becomes available to plants.

2.2.4 **Soluble Potassium**

Only a very small proportion of the total soil K is dissolved in the soil water at any one time (McLean and Watson, 1985). In New Zealand soils with a mean annual rainfall greater than 1000 mm the concentration varies from 3 to 30 $\mu g\ ml^{-1}$ K (During, 1984). Assuming that plants extract K only from solution, the amount of K in the soil solution is approximately 5-15 kg K ha^{-1} in the top 15 cm of soil (assuming a K^+ concentration of 3-30 $\mu g\ ml^{-1}$ and a mean water content of 30%) and is usually an inadequate amount for plant growth over an entire growing season (pasture plants recover 280-430 kg K ha^{-1} $year^{-1}$; section 2.3.4). Obviously replenishment of solution K^+ must occur from other forms of soil K or from returns via the cycle, such as excreta returned to the soil.

2.2.5 **Transformations of Soil Potassium**

Potassium can move from one soil form to another whenever the equilibrium that exists between the forms (Figure 2.2) is disturbed through removal or addition of K. Rates of transformation vary between the forms from fast to very slow (Selim et al., 1976). For example, exchangeable K^+ equilibrates with soil solution within one hour on most soils (Barber, 1984).

Talibudeen et al. (1978) examined the desorption of K^+ from a soil containing 12,000 mg K kg^{-1} (mainly as illite) using Ca-saturated resins as sinks for solution K^+. From these results they postulated that during continuous K removal by plant growth the concentration of K^+ in the soil solution bathing the plant roots is likely to be $<10^{-5}$M. The rate of release of K^+ from exchangeable K^+ and nonexchangeable K^+ in equilibrium with a solution of this K^+ concentration would mean that the reserves of exchangeable K^+ and nonexchangeable K^+ would continue to release K for 35 days and 100 years respectively.

Movement of K^+ into nonexchangeable positions in clay minerals has been described as fixation and can be considered to be the reverse process to weathering (Rich, 1968). The amount of fixation which can occur within a soil depends on the type of clay mineral and the degree of K^+ depletion. It does not occur with organic matter, kaolinite, chlorite or mica, is slight in smectite, variable in illite and most pronounced in vermiculite (Doll and Lucas, 1973).

Table 2.1 Total and exchangeable K content (me $100g^{-1}$) of major dairy farming soils in New Zealand (data from Metson, 1980).

Soil group	Total K	Exchangeable K
Yellow-brown loams		
Waikato region	18	0.8
Taranaki region	34	1.7
Yellow-brown pumice soils	43	0.5
Central yellow-grey earths	31	0.4
Central yellow-brown earths	30	0.5
Central yellow-brown sands	25	0.1
Recent soils	45	0.9

Potassium fixation capacities have been measured between 0.3 (for soil containing unweathered mica) and 0.6 me K g^{-1} clay (for soil containing expanded 2:1 micas; Arifin et al., 1973).

Since fixed K may be more available to plants than structural K (Rich, 1968) fixation is considered to be beneficial as it reduces potential leaching losses of K and limits luxury uptake by plants, particularly after deposition of K in urine, dung or fertiliser on to the soil.

2.2.6 Conclusions

The nature of soil K is well understood. The literature indicates that the ability of the soil to provide both a source and a sink of plant available K^+ in the K cycle is dependent on the type of clay mineral present. Where 2:1 clay minerals and allophane are present, the soil is able to retain K as exchangeable and nonexchangeable K^+. In soils which lack these clay minerals there is little in the way of a K source and a poor ability to retain K. Therefore the type of minerals present in the soil will affect both the amount of K fertiliser required for plant growth and their ability to retain K against leaching.

2.3 PLANT POTASSIUM

Potassium exists in an inorganic form in plant cells where it functions in several physiological processes including the maintenance of electrical balance at membranes, enzyme activation, carbohydrate production and transport, and stomatal activity (Lanyon and Smith, 1985).

The amount of K in the plant compartment, which comprises the K in the herbage and in the roots, is a function of the dry matter (DM) yield and the tissue K concentration. The plant compartment represents the route by which soil K reaches the above ground portion of the cycle. To predict the amount of K that is transferred to the animal compartment, accurate estimates of pasture growth rates and K herbage concentration are required.

2.3.1 Pasture Dry Matter Yield

Data on annual pasture production collected from many sites throughout New Zealand shows that considerable year to year variation in annual production can occur due to differences in climatic conditions, particularly temperature, moisture and light (Baars, 1982). Despite this the mean annual production from sites in the major dairying regions is fairly constant from year to year and most sites are capable of producing 11,000 kg dry matter (DM) ha^{-1}, as measured by monthly mowing trials (Baars, 1982). Grazing management can also affect pasture production by controlling the balance between the amount of green leaf available for photosynthesis and the amount of senescent material present (Leaver, 1987). On intensively managed dairy farms annual pasture production may be as high as 17,000-19,000 kg DM ha^{-1} (Bryant, 1982).

The typical seasonal pattern of pasture growth for the North Island of New Zealand shows highest growth rates in the spring (usually October) of 70 kg DM ha^{-1} day^{-1}, and lowest growth rates of 10-20 kg DM ha^{-1} day^{-1} over the winter months of June and July (Baars, 1982). Using data from the Waikato and Hawkes Bay regions, Baars and Waller (1979) identified the limiting climatic factors on pasture growth throughout the year. They found that in general soil moisture limited growth over summer and autumn (January to April); growth from March to October was limited by soil and air temperatures; during November and December climate had little influence on pasture growth as the grass species change from a vegetative to reproductive state.

2.3.2 Concentration of Potassium in Pasture

Like pasture DM yield, the K concentration of the herbage can vary from one site to another and also from one season to another. A considerable range of K concentrations from 0.98 to 5.21% have been reported for samples of mixed herbage from New Zealand pastures (Smith and Middleton, 1978).

Research into the effect of plant K concentration on the growth of the two main pasture species, perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*), has suggested that there are critical plant K concentrations for these two species of 2.2 and 2.0% respectively (McNaught, 1970). If the K concentration is below this critical level then a reduction in pasture yield may be expected. For pastures on dairy farms which have a high N status (due to topdressing with N fertiliser or through heavy applications of potassic superphosphate fertilisers which have encouraged a high level of N fixation by clovers) the critical level for ryegrass may be higher at 2.8% (G.S. Smith et al., 1985).

The wide range in pasture K concentrations reported by Smith and Middleton (1978) reflects the variation in plant uptake of K that can occur. Some of the main factors influencing K uptake are discussed in section 2.6.1.

2.3.3 Root Potassium

The weight of roots under established pasture in New Zealand has been measured at 7,000-10,700 kg DM ha^{-1} to a depth of 200 mm (Campbell, 1945; Gibbs, 1986). Like the aerial proportion of the plant, root growth in perennial ryegrass in the Manawatu region of New Zealand follows a seasonal cycle with new roots initiated between March and October (Jacques, 1956; Caradus and Evans, 1977) and maximum growth occurring in the spring (Jacques, 1943; Caradus and Evans, 1977). While most of the root growth occurs near the soil surface between May and August (Caradus and Evans, 1977), as the roots become older they grow deeper down into the soil until a maximum depth is reached over the summer period (Jacques, 1956).

Very little research appears to have been carried out on the size of the K pool in pasture roots. Root weight estimates have been published of 11,210 kg DM ha^{-1} (to a depth of 600 mm) containing 1.5% K for a hypothetical cycle in Georgia, USA (Wilkinson and Lowrey, 1973), and 8,000 kg DM ha^{-1} (depth not specified) containing 0.5% K for a typical Waikato dairy farm (Middleton and Smith, 1978). The latter gives an approximate root pool size of 40 kg K ha^{-1}.

2.3.4 Conclusions

It is apparent from this review that the size of the plant K compartment can vary from year to year mainly through the influence of climatic factors particularly moisture and temperature. If a pasture annually produces between 11,000 and 17,000 kg DM ha^{-1} and has a mean K concentration of 2.8% then the amount of K flowing through the plant compartment will be between 300 and 480 kg K ha^{-1} yr^{-1}.

2.4 ANIMAL POTASSIUM

The grazing dairy cow has a major impact on the K cycle. As shown in Figure 2.1, of the K ingested, some is lost in animal products and in transfer to unproductive areas of the farm, while the remainder is redistributed around the paddock in discrete highly concentrated deposits of urine and dung.

Although K is essential as an intracellular cation and is one of the most abundant mineral elements in the animal tissue (Towers, 1983), only a small proportion (<4%) of the K ingested by the dairy cow is retained in animal tissue (Maynard and Loosli, 1962; Hutton et al., 1967; Paquay et al., 1969). Of the remainder of the ingested K, some is converted in to animal products such as milk, but the majority (90-92%) is excreted (Sears, 1950; Hutton et al., 1967). Most of the K in plant material is water soluble, and readily absorbed by the small intestine. Hence, the majority (88-90%) of the unutilised K is excreted in urine with the remaining 10-12% in the dung (Maynard and Loosli, 1962; Hutton et al., 1967).

The average New Zealand dairy cow derives all her K intake from pasture, either *in situ* or as hay or silage. Supplementary feeds such as grain or concentrates are used on only a few farms, and where used they comprise only a small proportion of the diet.

The amount of K ingested by the dairy cow is dependent on the amount of pasture available, the proportion of this that is ingested and the K concentration of the pasture. On most dairy farms the pasture is rationed out to the cows using a rotational grazing system. Utilisation of this pasture will vary according to stocking rate.

The amount of K that is annually ingested by a dairy cow can be estimated from the cow's milkfat production and the conversion factor of 25 kg of pasture DM for each kg milkfat produced (Bryant, 1982). For a cow that produces 150 kg milkfat yr^{-1} this will equate to an intake of 3750 kg DM or 105 kg K cow^{-1} (assuming the DM has a 2.8% K concentration). For a stocking rate of 3 cows ha^{-1} this represents a flow through the animal compartment of 315 kg K ha^{-1} yr^{-1}. A comparison of this value with the amount of K passing through the plant compartment (section 2.3.4) indicates that at this stocking rate ingestion of plant K is high (approximately 66-100% utilisation).

2.4.1 Conclusions

Grazing dairy cows are capable of ingesting high quantities of K but they require only a small amount (<10%) of the K they ingest. The remainder is voided on to the soil in small highly concentrated areas of dung and urine. There may also be a transfer of excreta to unproductive areas of the farm such as raceways and the milking shed which will result in a loss of K from the productive areas of the farm. Due to the large amount of K that is excreted and the mobile nature of this compartment, the subsequent redistribution of animal K by the cow is an important part of the K cycle. A more detailed review of the fate of animal K is given in section 2.6.3.

2.5 POTASSIUM GAINS

The amount of K in the soil-plant-animal cycle is increased through the application of fertiliser and the addition of K in rainfall and irrigation water. Inputs of K through these pathways can alter the size of the soil K compartment and therefore the amount of K that moves from the soil to other compartments.

2.5.1 Potassium Fertiliser

The fertiliser requirements of New Zealand's dairy pastures varies according to the amount and type of clay mineral present in the soil (section 2.2).

Many field trials have been conducted by the New Zealand Ministry of Agriculture and Fisheries (MAF) to investigate how the responsiveness of plant growth to K fertiliser varies with soil type. Initially these were simple observational trials (Woodcock, 1936) but in later trials measurements of pasture production, and chemical analysis of herbage and soil were included (e.g., Thomson, 1982; Williams et al., 1986). These trials have been useful in identifying soil types that are likely to respond to K fertiliser.

Most of the trial designs in these experiments included a range of application rates of K in order to predict the fertiliser rate that would maintain 90% of maximum pasture yield (the maintenance rate of fertiliser) but did not involve grazing animals. In the absence of the grazing animal the results of these trials may not provide a reliable estimate of the maintenance rate of K fertiliser. The grazing animal can affect sward growth through the rapid return of nutrients in excreta, and by selective grazing and trampling (Wolton, 1963). For example, on the same soil type, the rate of maintenance K in a mowing trial was seven times greater than the maintenance rate determined in an intensively managed trial under sheep grazing, even though a high proportion of the clippings were returned to the mowing trial (Morton, 1981). The reason for this difference in responsiveness to fertiliser is thought to be due to a more rapid recycling of K and N via excreta in the sheep grazing trial compared to the mowing with clippings returned trial. Such results show that care must be taken when extrapolating results from small plot mowing trials to the grazed situation.

There have been very few experiments under grazing which are able to provide information on maintenance K requirements for dairy farms. The results from five field trials on intensively managed dairy farms producing 400-560 kg milkfat ha^{-1} yr^{-1} on yellow-brown loam soils in the Waikato region have shown that a maintenance rate of 70 kg K ha^{-1} yr^{-1} is required (O'Connor and Feyter, 1980). In general, annual maintenance requirements of 60 to 70 kg K ha^{-1} have been proposed by During (1984) for soils with a medium or high K retention that produce 9,000 to 14,000 kg DM ha^{-1} yr^{-1}.

The residual effect of past K fertiliser appears to last only one year on intensively managed dairy farms (O'Connor and Feyter, 1980) possibly due to the high amount of K lost through the grazing animals on these intensive farms (section 2.7.3). The results from this trial suggest that K fertiliser should be applied in the early spring to ensure that it is present for the peak pasture growth in spring/early summer (During, 1984).

2.5.1.1 *Form of potassium fertiliser*

In New Zealand the most common form of K fertiliser used is potassium chloride (KCl), which accounted for 99% of the total amount of K fertiliser imported into the country in 1986 (Ministry of Agriculture and Fisheries, 1987). Potassium sulphate accounted for the remaining 1% of the K fertiliser imported (Ministry of Agriculture and Fisheries, 1987). Due to its high cost (imported cost \$526 t^{-1} compared with \$195 t^{-1} for KCl; Ministry of Agriculture and Fisheries, 1987), potassium sulphate is used mainly for horticulture crops which are Cl^- sensitive, e.g., tobacco and lettuce (During, 1984).

Since Cl^- is the major anion applied in K fertilisers, leaching losses of K^+ may be enhanced as Cl^- is not adsorbed by soil minerals (Johnson et al., 1986; Mengel and Kirkby, 1987).

2.5.2 **Potassium in Rainfall and Irrigation**

The amount of K that enters the K cycle via rainfall and irrigation per annum is considered to be "negligible" (Campkin, 1985) compared with the total amount of K in the top 30 cm of the soil profile (2.5-55 t K ha^{-1}; During, 1984). Published estimates of K in the rainfall at sites which were less than 50 km from the western coastline of New Zealand include, 1.5 kg K ha^{-1} yr^{-1} (Campbell, cited in Lee et al., 1979), 5 kg K ha^{-1} yr^{-1} (McColl and Gibson, 1979b) and 1.4 kg K ha^{-1} for 6.5 months over the winter (Smith, 1979).

2.5.3 **Conclusions**

The main gain to the K cycle is via fertiliser. While there have been many field trials carried out to determine the appropriate rate of fertiliser for pasture production, most trials were carried out in the absence of the grazing animal and so provide little information on the fertiliser requirements of a grazed pasture. The potential for predicting the amount of fertiliser required for dairy farms from a knowledge of the K cycle and the losses and gains of K that occur within the cycle will be discussed further in section 2.8.

2.6 TRANSFORMATIONS BETWEEN COMPARTMENTS

The rate of transformation of K from one compartment to another will determine the rate at which K cycles through the grazed pasture, which in turn partly affects the availability of K for plant growth. A knowledge of the rates of transfer from one compartment to another will serve to identify particular transfers which limit the rate of K cycling and control the losses of K from the cycle.

2.6.1 Plant Uptake of Soil Potassium

As mentioned in section 2.3, the amount of K in the plant compartment is highly variable due to a spatial and temporal range of both DM production and K concentration. The concentration of K in the herbage depends partly upon reactions that occur in the soil-root interface zone. Potassium moves to this zone mainly by diffusion along a concentration gradient induced by the depletion of K from solution at the root surface (Barber, 1984).

The mechanisms affecting K uptake which operate at the soil-root interface have been the subject of much research. As a result mechanistic models have been developed which predict K uptake by plants. Barber (1984) has reviewed these models which show that K uptake depends on the configuration and growth rate of the root system, K uptake characteristics per length of root and the K supply characteristics of the soil. Sensitivity analyses of these parameters on soya beans (Silberbush and Barber, 1983) and corn (Barber, 1985) have shown that the major factors affecting K uptake are firstly root length and radii, and secondly the rate of K supply by the soil as measured by concentration in soil solution, soil buffering capacity and the diffusion coefficient.

Whereas most of the research on plant uptake has involved crop species, the research into pasture plant species is limited. A characteristic of most pastoral systems is that the majority of the plant roots are found in the surface layers of the soil profile (Williams, 1968; Caradus and Evans, 1977). Since the amount of K taken up by the roots of the pasture species is related to the proportion of the root mass that is present at any one depth (Ozanne et al., 1965), it has been shown in field experiments that the majority of the K taken up by plants is derived from the surface layers of the soil profile (Ozanne et al., 1965; Peterson and

Smith, 1973). In particular, in a field experiment in Western Australia Ozanne et al. (1965) found that the amount of ^{42}K recovered by ryegrass and clover species was 25 times more from ^{42}K injected at the 5 cm soil depth than the 35 cm depth. They attributed this greater relative uptake of ^{42}K from the 5 cm depth of soil to a greater amount of plant roots at the 5 cm depth compared with the 35 cm depth. The amount of exchangeable K^+ at each depth may also have influenced K uptake but no data on the amount of exchangeable K^+ were provided.

In New Zealand no direct measurements have been made on the depth of soil from which pasture plants extract K^+, but there are indirect measurements in the literature of extraction from depths down to 61 cm (Weeda, 1978; During and Campkin, 1980). Evidence for this deep extraction is from a decline in soil exchangeable K^+ at depths down to 61 cm during the experiment, and from an apparent gain in plant K that can not be explained by analyses of soil above this depth.

To enable better quantification of measurements of plant available soil K^+ and leaching losses, further consideration needs to be given to defining the depth of the soil profile from which the majority of plant K is derived.

The amount of K^+ taken up by the herbage varies according to the type of plant species present in the sward, the season of the year and the amount of K^+ that is present in the soil solution. The effect of these factors on the amount of K which flows through the plant compartment and the ways of measuring the size of the plant available soil K pool will be discussed in the following sections.

2.6.1.1 *Effect of species on plant uptake*

New Zealand dairy pastures are mainly based on a grass-legume association of ryegrass and white clover. In these pastures it is common for there to be a greater K concentration in the ryegrass plants compared with the white clover (Melville and Sears, 1953; McNaught, 1958; McNaught and Dorofaeff, 1968; During, 1984). The reasons for the greater K concentration in the ryegrass plants have been attributed to a stronger competitive ability of the ryegrass plants due to the denser, more fibrous root system (Haynes, 1980; Mengel and Steffens, 1985), and the need to absorb cations to balance uptake of NO_3^- (Barber, 1984). Since clover plants obtain some of their N from nitrifying rhizobia in the form of ammonia and organic N (Mengel and Kirkby, 1987) they do not have the same requirement for cations. Further evidence for the grasses requirement for K to balance NO_3^- uptake is provided by

Carran (1988), who found that K^+ uptake by pasture grass species was significantly higher in soil where nitrification was occurring (i.e., NO_3^- uptake by grass species was high) than when NH_4^+ was the dominant form of plant available soil N.

2.6.1.2 *Effect of season on plant potassium uptake*

Seasonal variations in K concentration have been measured in pasture samples collected from different geographical areas in New Zealand (Hudson et al., 1933; Melville and Sears, 1953; McNaught and Dorofaeff, 1968; Metson and Saunders, 1978). These sites all showed a similar trend with a lower K concentration in the drier months of the year (summer) and higher levels in the wetter months particularly the early spring. Such an effect is mostly linked to soil moisture (Thomas and Hipp, 1968) because the diffusion of K^+ to the roots is greater at higher soil moisture contents (Barber, 1984; Kuchenbuch et al., 1986) when more of the soil's porosity is water filled and the tortuosity of the diffusion pathway is decreased (Barber, 1984).

This seasonal variation in K concentration was not observed by Reay and Marsh (1976) in a field trial in which perennial ryegrass and red clover were grown separately. They, however, selectively sampled only immature leaves whereas the other authors mentioned earlier in this section analysed plant material that varied in age. Since K in plants is mobile and is translocated from mature to young leaves (McNaught, 1970), by sampling only immature leaves Reay and Marsh may not have measured the K status of the whole plant.

2.6.1.3 *Effect of soil potassium status on plant potassium concentration*

Where the supply of soil solution K^+ to the roots is in abundance plant uptake occurs readily, which can result in plant K concentrations above the critical value (luxury consumption). In the survey carried out by Smith and Middleton (1978) on samples of mixed pasture species, 75% of the 2767 sites sampled had excessively high plant K concentrations (i.e., >2.5% as in this paper 2.0-2.5% K was considered to be adequate for plant growth). The authors suggested that these high values were due to heavy applications of fertiliser to the soil. Luxury consumption by pasture plants may be partly overcome through the use of slow release K fertilisers such as sulphur (S) coated potassium chloride (Saunders et al., 1988).

The presence of high levels of K^+ in the soil can decrease the plant uptake of other ions particularly Ca^{++}, Mg^{++} and Na^+ (McNaught et al., 1973; Smith et al., 1983; Barber, 1984; G.S. Smith et al., 1985). If this results in herbage with a high K/(Ca,Mg,Na) ratio metabolic problems in dairy cows can occur (Metson et al., 1966; Smith et al., 1983; Towers, 1983).

While plant Ca, Mg and Na concentrations are affected by soil K the reverse is not true (Barber, 1984). The only major element that seems to affect plant K uptake is N. At high plant N concentrations (e.g., 3.2% kjeldahl N and 500 $\mu g\ g^{-1}$ NO_3^--N) high K concentrations are required (2.8%) for maximum growth (G.S. Smith et al., 1985).

2.6.1.4 *Measurement of plant available potassium*

Soil test methods used to assess the quantity of soil K that is potentially available for plant uptake are mainly carried out by treating the soil with an extractant. The quantity of K extracted depends on the type of extractant used. For example, extractants that contain NH_4^+ ions will extract mainly exchangeable K^+, while acid extractants release a proportion of nonexchangeable K^+ (Metson, 1980; McLean and Watson, 1985). These techniques provide a relative estimate of the quantity of plant available K^+, but they provide little information on the rate at which the soil supplies K^+ to the plants. Evidence that the rate of supply needs to be considered for New Zealand soils comes from experiments such as that of During and Duganzich (1979). This experiment showed that there was a highly significant relationship (r^2=0.9) between K uptake by white clover and both K concentration and soil K buffering capacity on soils with different CECs.

An alternative method of measuring the quantity of plant available exchangeable and nonexchangeable K^+ in soils is to determine the ability of a soil to maintain a constant K^+ concentration, or intensity, in the soil solution despite additions or removals of solution K^+. This concept of a Quantity/Intensity relationship for soils has been developed for K by Beckett (1964a, 1964b) based on the ratio laws of cation exchange proposed by Schofield (1955). Other workers have used the free energy of exchange (Woodruff, 1955) to describe K availability. A comprehensive review of these methods has been written by Lee (1973). More recently electroultrafiltration has been promoted as a method of determining the concentration and rate of K^+ supply of soils (Nemeth, 1972).

While these methods of measuring K^+ intensity in soils have received considerable interest, the analyses are laborious to carry out and so most researchers involved in predicting plant available soil K^+ prefer to use quantity measurements (Metson, 1980).

In New Zealand the method adopted by MAF for measuring plant available K is the Quick test (QTK). This test provides an estimate of the quantity of K that is immediately available for plant uptake by extracting water soluble and some exchangeable K. The QTK involves shaking the air dried soil with 1 M ammonium acetate for 2 minutes (Cornforth, 1980). One problem with determining the plant available soil K status of a grazed pasture is the high degree of variation in exchangeable K^+ (D. Wheeler, pers. comm.) caused by the uneven distribution of dung and urine. Due partly to this spatial variation, and partly to the short equilibration time, which on average removes 70% of the exchangeable K^+ normally extracted by complete exchange with 1 M NH_4^+ at pH 7 (Cornforth and Sinclair, 1984), the QTK has an expected error of 30% (I.S. Cornforth, pers. comm.).

Measurement of the amount of nonexchangeable K^+ that may become plant available is commonly carried out using boiling 1 M nitric acid (McLean and Watson, 1985). Several successive extractions with boiling nitric acid have been used by Metson (1968) to estimate "Kc", the long term supply of K from nonexchangeable sources in the soil. Since the Kc value is unaffected by short term additions and removals of soil K it reflects the soil's mineralogy and degree of soil development (Metson, 1980). The Kc test has been shown to be poorly correlated with the plant uptake of soil K (Jackson, 1985).

Fertiliser recommendations made by New Zealand's MAF Computerised Fertiliser Advisory Service (CFAS) model are based on both the QTK and Kc tests (Cornforth and Sinclair, 1984). There is, however, some evidence to suggest that on New Zealand soils with high levels of nonexchangeable but plant available K^+ (e.g., on recent soils which contain 2:1 clay minerals such as illite), the CFAS model may overestimate K fertiliser requirements (Williams et al., 1986) because the QTK does not measure this potential source of plant available K^+ (During and Campkin, 1980; Williams et al., 1986). In this situation a measure of nonexchangeable K^+ that will become plant available in the short term needs to be made. While many extracts have been used to measure nonexchangeable soil K^+ (Doll and Lucas, 1972; Metson, 1980) few of these tests can be used as a rapid and reproducible routine test for predicting fertiliser requirements (Doll and Lucas, 1972). One soil test which has the potential to overcome these problems is the sodium tetraphenylboron extraction (TBK) which removes solution, exchangeable and some nonexchangeable K^+ from soil (Jackson, 1985). The TBK extraction is currently used in the CFAS K model on recent soils to predict their plant available soil K^+ content (Cornforth and Sinclair, 1984).

In New Zealand, During (1973) used activity ratios to determine the effective relative affinity of soils for K^+ and Mg^{++} retention. Using these ratios he was able to distinguish between soil groups that have low, medium or high K^+ retention abilities. The results showed that K^+ retention was related to the type of clay mineral present (e.g., soils containing vermiculite had the highest K^+ retention, while soils containing quartz as the dominant mineral had the lowest retention). Most soils have medium or high retention with only a few showing low retention due to a lack of 2:1 clay minerals (e.g., gley podzols and some coarse textured yellow-brown pumice soils; During, 1984).

The amount of K in the plant available soil pool depends on the concentration of available soil K^+ and the activity of plant roots at each soil depth. There appears to be little evidence on which to determine what depth of soil the plants derive the majority of their K^+ from. The MAF soil tests are conducted on a soil sample collected from the top 7.5 cm of the soil profile. However, it has been suggested by Weeda (1978) and During and Campkin (1980) that in New Zealand K uptake by pastures may occur to depths of 61 cm below the soil surface. As already discussed (section 2.6.1) the depth of soil from which pasture plants recover the majority of their K needs to be defined in order to quantify the size of the soil compartment that is available for plant uptake.

2.6.2 **Return of Plant Potassium to the Soil**

Of the herbage K that is not ingested by the grazing animals a proportion will be returned to the soil compartment through leaf leaching and decomposition of plant and root material.

2.6.2.1 *Leaf leaching*

The leaching of K from leaves by the action of rain, dew, mist and fog has been shown to occur in a wide range of plant species including grasses (Tukey, 1970). In New Zealand, measurements of K leaching from leaves have been made for trees only (Miller, 1963). The amount of K leached from pasture plants has been measured in England by Clement et al. (1972). Their results from pot experiments showed that 1.5 kg K ha^{-1} could be leached from a single heavy rain storm (15 mm in 1 hour) or 8 hours of continuous light rain (1.25 mm $hour^{-1}$). The authors suggested that in a year 100 kg K ha^{-1} may be leached from a pasture sward. However, since this K is readily reabsorbed by the roots, K from foliar leaching is largely recycled and therefore is not necessarily a loss from the soil-plant cycle (Clement et al., 1972).

2.6.2.2 *Plant and root decomposition*

Grazing animals will reject pasture because of poor positional availability (e.g., low sward height), poor quality due to a high proportion of stem and dead matter, and contamination by excreta or soil through trampling. The amount of herbage that is rejected is inversely related to grazing pressure. As grazing pressure and therefore pasture utilisation increases, more of the plant K is cycled through the animal at each grazing and a smaller amount is left behind in the plant pool. In general, New Zealand dairy farms are managed intensively so the amount of litter material in dairy pastures is usually small and only a small amount of K will be returned to the soil in this manner each year.

The majority (70%) of the K in plant material is water soluble, and the remainder is readily released by microbial action (Alexander, 1977), so the K in any senescent material left behind by the grazing animal will be rapidly returned to the soil as water soluble K^+.

The proportion of the root system that decomposes each year depends on the environmental conditions but may be as high as 100% in perennial ryegrass (Jacques, 1956; Williams, 1968; Troughton, 1981). More recent research in Canterbury has shown very high rates of turnover of ryegrass roots, with autumn produced roots living for 144 days and spring/summer produced roots living for only 29-81 days (Gibbs, 1986). As the roots decay the K may be either reutilised in the plant or returned to the soluble K fraction of the soil pool.

2.6.3 Return of Excretal Potassium to the Soil Pool

Ninety percent of the K ingested by the dairy cows is excreted in dung and urine (section 2.4). Unfortunately this return is in a nonuniform distribution pattern which makes it difficult for excretal K to be recycled completely through subsequent plant uptake. The amount of K excreted, the availability of this K for plant uptake, and the distribution pattern of excreta on the soil will affect the extent to which K is recycled. The amount of K in excreta and the distribution pattern of excreta deposition will be discussed in this section. The availability of excretal K for plant uptake will be reviewed in section 2.6.4.

The amount of K excreted in dung and urine is related to the K concentration of the ingested herbage and also to the climatic conditions. The distribution pattern of the excreta is dependent on the frequency, amount and surface area covered by each excretion. Any difference in the distribution between day and night time periods will also affect the recycling of excretal K.

All of the K in urine and 97-100% of the K in dung is considered to be water soluble (Davies et al., 1962; Bear, 1965; Barrow, 1967).

2.6.3.1 *Potassium concentration in excreta*

The literature reports a considerable range of K concentrations in dairy cow urine and dung from 0.35 to 1.56% (Table 2.2). This variation in K content can be attributed to several factors including differences between individual cows, and daily and seasonal fluctuations caused by diet and temperature. The level of excretal K has been found to be correlated with dietary K, suggesting that as K intake increases excretion of K will also increase (Hutton et al., 1967; St Omer and Roberts, 1967; Paquay et al., 1969).

Larvor and Brochart (1959) and Betteridge et al. (1986) have shown that there is a diurnal variation in the K concentration of urine, which is higher at night and lower during the day. Bovine animals rely on urination as a mechanism for controlling body temperature. On a hot day they urinate more frequently, which results in a lower K concentration compared with a cold day (Betteridge et al., 1986).

2.6.3.2 *Number of excretions per day*

Mature cattle urinate 8-12 times per day and defaecate 11-16 times per day (Table 2.3). The mean values presented in Table 2.3 show little variation in the number of daily urinations and defaecations. Such consistency is surprising as the number of daily excretions can be influenced by a number of factors including the season of the year (MacDiarmid and Watkin, 1972b) and the weather (Betteridge et al., 1986). For example, in a trial which monitored the urination pattern of two beef steers over a time period of 24 hours, Betteridge et al. (1986) found that the number of urinations per 24 hours varied between 13 and 73, the higher number being associated with very high day time temperatures.

While most of the research has been concerned with adult cattle no measurements appear to have been made for younger stock (heifers and calves) which are also involved in cycling K.

2.6.3.3 *Size of each excretion*

Each urination event by dairy cows has a mean volume of 1.6-2.2 dm^3 and each defaecation has a mean wet weight of 1.48-2.27 kg (Table 2.4). These mean values are subject to large variation. For example, Doak (1952) recorded a range of urination volumes of 0.85-2.85 dm^3. Other authors have recorded a range of daily urine volumes from 11-49 dm^3

Table 2.2 Concentration of K in dairy cow urine (% of wet weight) and dung (% of dry weight).

Reference	% K
Urine	
Petersen et al. (1956b)	0.95
Saunders & Metson (1959)	0.55
During & McNaught (1961)	0.35-1.08
Mundy (1961)	0.59-1.56
Davies et al. (1962)	0.78
Hutton et al. (1967)	0.5-1.3
Hogg (1968)	0.78
Robertson (1972)	0.64
Richards & Wolton (1976)	0.41
Hogg (1981)	0.63
Ledgard et al. (1982)	0.85-1.04
Safley et al. (1984)	0.75
Dung	
Hutton et al. (1967)	0.4-1.4
During & Weeda (1973)	0.86-1.51
Weeda (1977)	1.43
Hogg (1981)	0.91

Table 2.3 Mean number of urinations and defaecations per 24 hours by cattle.

Reference	Urinations	Defaecations	Type of stock
Johnstone-Wallace & Kennedy (1944)	8.5	11.75	beef cows
Castle et al. (1950)	9.8	11.6	dairy cows
Hancock (1950)	10.1	12.2	dairy cows
Goodall (1951)	11	12	dairy cows
Hardison et al. (1956)	9.4	15.4	dairy cows
Petersen et al. (1956a)	8	12	dairy cows
MacLusky (1960)		11.6	dairy cows
Davies et al. (1962)	10	12	dairy cows
Wardrop (1963)	12.1	16.1	dairy cows
Weeda (1967)		10.5	beef steers
MacDiarmid & Watkin (1972b)		13.9	dairy cows
Robertson (1972)	10		dairy cows
Richards (1973) (cited in Morton 1984)		14.6	dairy cows
Weeda (1979)		10.5	beef steers

Table 2.4 Mean volume of a single urination and wet weight of a single defaecation for cows.

Reference	Urination (dm^3)	Defaecation (kg wet wt)	Type of stock
Johnstone-Wallace & Kennedy (1944)		1.77	beef cows
Goodall (1951)		1.48	dairy cows
Waite et al. (1951)		2.27	dairy cows
Doak (1952)	1.6		dairy cows
Davies et al. (1962)	2.2		dairy cows
MacDiarmid & Watkin (1972b)		1.82	dairy cows
Robertson (1972)	2		dairy cows

(Hutton et al., 1965), 1-33 dm^3 (Paquay et al., 1970a), 5-34 dm^3 (Paquay et al., 1970b) and 5.8-54.7 dm^3 (Betteridge et al., 1986). Measurements for individual defaecations have shown a range of wet weights between 0.45 and 6.79 kg (Goodall, 1951).

This variation in excretal output can be attributed to various factors including the time of the day (with larger excretions in the early morning; Goodall, 1951; Betteridge et al., 1986), the quantity of water ingested (Paquay et al., 1970a; 1970b) and variations in air temperatures (Betteridge et al., 1986).

2.6.3.4 *Ground surface area covered by each excretion*

The ground surface area covered by a single defaecation ranges from 0.05 to 0.09 m^2 (Table 2.5). There is a considerable range in the area of urine patches recorded in the literature, from 0.13 to 3.52 m^2 (Table 2.5). One reason for this variation in urine area is that some measurements are of areas wetted by the urine (e.g., Petersen et al., 1956a; Davies et al., 1962), while others are for measurements of the areas of pasture that are visibly affected by the urine. The affected area tends to be larger than the actual wetted area (Doak, 1952; Lotero et al., 1966).

Other factors causing a high variation in the measured area covered by a urine event may be wind, slope and soil physical conditions. During and McNaught (1961) found that urine events simulated using 2 dm^3 of water containing chalk as a marker, covered an area of 0.26 m^2 on a calm day, 0.50 m^2 on a windy day and 0.65 m^2 on a sloping site.

2.6.3.5 *Total ground surface area covered by excreta*

The proportion of the paddock that is covered by excreta (P) during a period of time can be calculated from the following formulae:-

$$P = 100 * D$$
$$D = N * a / A$$

where D is the mean density of excreta deposited on the paddock during the time period; N is the total number of excretions on the paddock that occur during the time period; a is the total ground surface area covered by each individual excretion; and A is the total area of the paddock (Petersen et al., 1956a). This distribution assumes that until the entire paddock has been covered once by excreta no point is covered a second time and so on. MacLusky (1960) used this approach to predict that after an accumulated stocking rate of 1000 cow days ha^{-1} 50% of the pasture would have been affected by dung. However, measurements

Table 2.5 Ground surface area covered by individual urinations and defaecations.

Reference	Area (m^2)	Type of stock
Urine		
Petersen et al. (1956a)	0.28	dairy cows
Davies et at. (1962)	0.19	dairy cows
Lotero et al. (1966)	0.89-1.27	beef cows
Hogg (1968)	0.18	dairy cows
Richards (1973) (cited in Morton 1984)	0.13-3.52	INA[1]
Wilkinson & Lowrey (1973)	0.52-1.33	INA
Richards & Wolton (1976)	0.49	dairy cows & steers
Dung		
Johnstone-Wallace & Kennedy (1944)	0.06	beef cows
Petersen et al. (1956a)	0.09	dairy cows
MacLusky (1960)	0.05	dairy cows
Davies et al. (1962)	0.07	dairy cows
MacDiarmid & Watkin (1972b)	0.07	dairy cows
During & Weeda (1973)	0.05	beef steers
Wilkinson & Lowrey (1973)	0.05	INA
Richards & Wolton (1976)	0.05	dairy cows & steers

[1] INA information not available.

reported by Marsh and Campling (1970) have shown that after this stocking rate only 15% of the pasture was affected by dung. Therefore the overlapping of excreta patches that occurs in the field under high stocking rates can have a significant effect on the proportion of pasture that is covered by excreta (Petersen et al., 1956a; Richards and Wolton, 1976).

To allow for excreta overlap and for the nonuniform distribution of excreta that occurs by grazing animals, the negative binomial distribution has been used to predict P, the proportion of the paddock that is covered by excreta during a period of time (Petersen et al., 1956a; Richards and Wolton, 1976). The negative binomial distribution calculates P from the following formula, where D is the same as above, (Bliss and Fisher, 1953):-

$$P = 100 - 100/[(D+\mathrm{k})/\mathrm{k}]^{\mathrm{k}}$$

The mean value of k has been derived as 1.5 for a low stock density of 58 cow days ha^{-1} (Petersen et al., 1956a). For higher stock densities which resemble New Zealand dairy farms, a value of 7 has been derived for k, which suggests a more uniform distribution of excreta throughout the paddock due to a lack of opportunity for camping (Richards and Wolton, 1976).

Most of the measurements used to test the accuracy of these models have been based on dung distribution, since it is easy to measure in the field. However, Petersen et al. (1956a) observed that there was no difference between the distribution of dung and urine, therefore it should be possible to use these models to predict the area covered by urine as well as dung.

2.6.3.6 *Distribution of excretions between day and night*

Observations made on the number of daily excretions have shown that there tends to be a higher proportion of defaecations and urinations occurring at night (between the evening and morning milkings) compared with during the day (between the morning and evening milkings; Castle et al., 1950; Waite et al., 1951; Hardison et al., 1956). Hancock (1950) found the opposite to this and suggested that the ratio of excretions deposited during the night and day periods was similar to the proportion of the time in each period. Measurements in New Zealand on the weight of excreta that is deposited during the night and day have shown that similar weights of dung and urine are deposited in both time periods (Goodall, 1951; Hancock and McArthur, 1951).

On farms where separate paddocks are grazed during the day and night periods, a transfer of fertility has often been observed from the day to the night paddocks. Hancock and McArthur (1951) have explained this transfer as being due to the cows ingesting 40% of their grazing at night while depositing 50% of the excreta there. This is often compounded by the tendency for night paddocks to be smaller than day paddocks and for night paddocks to be grazed more often than day paddocks.

2.6.4 Fate of Excretal Potassium After Deposition on the Soil

A knowledge of the subsequent fate of the excretal K once deposited on the soil is important for determining the rate at which K is returned to the plant and also for identifying losses of K that may occur from the cycle.

The subsequent fate of excretal K has been the subject of several research projects. The majority of this research has been carried out in New Zealand. Both dung and urine result in an increase in the growth rate of the affected pasture and an increase in the K concentration of the herbage (e.g., Saunders and Metson, 1959; During and McNaught, 1961; MacDiarmid and Watkin, 1971; During et al., 1973; Weeda, 1977; Hogg, 1981; Ledgard and Saunders, 1982; Ledgard et al., 1982; Saunders, 1984). The return of excretal K to soils involves application of very high rates of K (equivalent to 1000 kg K ha^{-1}; Saunders, 1984) to a relatively small area of soil, therefore it is unlikely that all of the excretal K will initially be recovered by the plants. Consequently the fate of excretal K due to retention by the soil and movement down through the soil profile via leaching needs to be considered.

2.6.4.1 *Urine potassium*

Research work on the fate of K from applications of dairy cow urine to soils have shown that between 2.5 and 90% is recovered by plants, 1 to 72% is leached and 0 to 33% is retained in the soil (Table 2.6). The high variation of these results is due to the variety of experimental conditions; the duration of the experiments, the number and size of leaching events, soil preparation (if any), and whether the experiments were conducted in the field, glasshouse or laboratory.

The duration of the effect of urine K on subsequent plant growth has also been estimated from the experiments in Table 2.6, therefore it is not surprising that there is a wide range of reported values from 6 months (Saunders and Metson, 1959; During, 1984) to "little more than one year" (During and McNaught, 1961). On dairy pastures with a high K fertiliser history the increased pasture growth observed around a urine patch is usually caused by a response to N (as urine contains up to 1.3% N; Hutton et al., 1967) rather than K. The response by the pasture to the N in urine is of a shorter duration than the response to K

Reference	Soil	Experimental Technique	% Leached	% Recovered by plants	% Remaining in soil	Duration of Experiment
New Zealand Experiments						
Saunders & Metson (1959)	yellow-brown loam	intact soil cores 375 mm rain applied	7 14	52 41	10 14	6 months
Davies & Hogg (1960)	yellow-brown loam	glasshouse pot trial 2500 mm rain applied	30	51	7	12 months
During & McNaught (1961)	yellow-grey earth	1.field trial 2.leaching column	NR 22	23	NR NR	25 months 8 weeks
Hogg (1968)	yellow-brown loam	1.field lysimeter 2.leaching column	2 8	90	0 NR	32 months 4 weeks
	yellow-brown sand	3.leaching column	72		NR	4 weeks
Hogg (1981)	yellow-brown sand	field lysimeter	4	51	33	4 years
Ledgard et al. (1982)	yellow-brown loam	field experiment	NR	15	NR	3 months
Overseas Experiment						
Pfitzenmeyer (1970)	sandy loam	field experiment	NR	15	NR	2.5 years

Table 2.6 Fate of urine potassium as a % of that applied in laboratory and field experiments.

(Lotero et al., 1966; During and McNaught, 1961). In general the length of the urine effect on herbage growth depends on the rate of plant uptake and the amount of K (and N) lost due to leaching. For example, Lotero et al. (1966) in the southeastern states of the United States of America, and Pfitzenmeyer (1970) in France, found that the duration of the herbage response to urine was much shorter from an autumn urination than from urinations in the spring and summer due to leaching over the winter.

Urine affected pasture is not normally rejected by grazing cows (Marsh and Campling, 1970) provided the pasture does not become too rank and therefore loses digestibility. There have even been reports of urine affected pastures being preferentially grazed by animals (Norman and Green, 1958; MacLusky, 1960) provided the pasture was not damaged due to urine burn (Richards and Wolton, 1975). Preferential grazing of urine patches would enhance rapid recycling of K.

2.6.4.2 *Dung potassium*

Potassium from dung has been shown to increase soil exchangeable K^+ within three days after application (MacDiarmid and Watkin, 1972a). Very little of the K in dung appears to be leached down through the soil profile (MacDiarmid and Watkin, 1972a; Weeda, 1977; During et al., 1973) and recoveries in the herbage of 80% (in 3.5 months) and 100% (in 3 years) of the K applied in dung have been recorded by During et al. (1973) and Weeda (1977) respectively.

Dung can affect nutrient cycling in two ways. Firstly, there is often some herbage death immediately beneath the dung patch (Weeda, 1967), particularly if the dung remains for more than 15 days before decomposition is complete (MacDiarmid and Watkin, 1971). These areas may remain bare for up to 2 years (Weeda, 1967). The rate of decomposition of the dung patch varies according to the season of the year and is particularly affected by the water content of the dung, climatic conditions and activity of soil dwelling invertebrates (Marsh and Campling, 1970).

Secondly, grazing cattle tend to reject herbage that is contaminated by dung. This topic has been well reviewed by Marsh and Campling (1970). However, under intensive grazing management such as occurs in dairy pastures, rejected herbage is likely to be kept to a minimum (Norman and Green, 1958; MacLusky, 1960; Weeda, 1967; Greenhalgh and Reid, 1969; Boswell, 1971; MacDiarmid and Watkin, 1972b).

2.6.5 Conclusions

Information on the uptake of soil K by plants and the size of the plant available soil K pool is limited by a lack of knowledge on the depth of soil from which plants recover the majority of their K requirements.

The review on the return of excretal K to the soil indicates that the grazing animal has a major impact on the redistribution and conservation of K. Information on the spatial distribution of excreta is well documented and it is possible to use this information to predict the total ground surface area of a paddock that will be covered by excreta during a particular time period. Since the majority of the excretal K that is returned to the soil is in the form of urine, e.g., of the 315 kg K ha^{-1} ingested by the cows (section 2.4) 256 kg K ha^{-1} is excreted in urine, the subsequent fate of this K due to plant uptake, soil adsorption and leaching is important. Evidence in the literature indicates that there is some confusion over the fate of urine K. In particular there is a lack of information on the redistribution of urine K due to plant uptake and leaching under field conditions. Further research is also required to determine what influence different soil factors such as CEC and clay mineralogy, have on adsorption and leaching of urine K.

2.7 POTASSIUM LOSSES

Losses of K from the cycle of a grazed pasture are inevitable and occur through leaching, removal of herbage K in hay and silage crops, removal in animal products and transfer of excretal K to unproductive areas of the farm. A knowledge of the amount of K lost via these mechanisms is important as this amount needs to be replaced to maintain productivity of the grazed pasture.

2.7.1 Loss of Soil Potassium Due to Leaching

Potassium can be leached through surface runoff or movement down through the soil profile beyond the plant rooting depth.

New Zealand dairy farms comprise mainly permanent pasture on flat to rolling land hence loss of K through surface runoff is likely to be minimal and associated only with heavy rain storms after fertiliser application or after grazing (McColl and Gibson, 1979a; Smith et al., 1984). For example, on a gley podzol under very high rainfall (2500 mm per annum) where 80% of the rainfall runs off the soil surface, 20 kg K ha^{-1} were lost in the first 5 months after application of 100 kg K ha^{-1} as fertiliser (Lee et al., 1979). On a lower rainfall site (1020 mm per annum) more characteristic of North Island dairy farms, where 38% of the rainfall was measured as runoff, 5 kg K ha^{-1} (representing 3.5% of the fertiliser K) were lost during the first 15 weeks after fertiliser application (Smith et al., 1984). Surface run-off water can contain both dissolved and particulate forms of K. The results of Lee et al. (1979) and Smith et al. (1984) were obtained from filtered (<45 μg) samples of run-off water and hence contained mainly dissolved K. However, if particulate losses of K were also considered, especially on soils which contain K rich clay minerals such as the yellow-grey earth soil used in the experiment of Smith et al. (1984), then the total amount of K lost would have been greater than the 5 kg K ha^{-1} reported.

Losses of K in percolating drainage waters have been investigated in many laboratory, glasshouse, lysimeter and field studies (e.g., Munson and Nelson, 1963). When considering leaching losses of K from an ecosystem by movement down through the soil profile, it is important to define the depth of soil beyond which little K is recovered by plants. Potassium leached beyond this depth can be considered to be lost from the immediate cycle. Unfortunately the majority of the published literature relating to leaching of K fails to define this depth. In New Zealand field studies leachates have been collected from soil depths of 75 cm (Smith et al., 1984) and 70 cm (Steele et al., 1984), however, the effective rooting depths of the pasture plants were not considered and these depths may have been sampled for other reasons than where effective plant rooting depth ceased (e.g., the presence of tile drains which facilitated the collection of leachate; Smith et al., 1984).

2.7.1.1 *Effect of Sol type, pH, rate and type of potassium fertiliser on potassium leaching*

The amount of K leached during laboratory and glasshouse experiments using sieved and repacked columns or pots of soil, has been shown to vary according to soil type, mainly due to differences in soil chemical properties. Of particular importance in determining the amount of K in the soil solution is the CEC. Leaching losses of K tend to be lower in soils which have a high clay and organic matter content and which contain weathered clay minerals such as illite, all of which increase soil CEC (Munson and Nelson, 1963). For example, in an experiment with leaching columns of sieved repacked soil, Davies et al. (1962) found that leaching losses of K were very low (4 to 6% of applied K) in soils containing micaceous clay minerals, but were significantly higher (28 to 69%) in soils that lacked these minerals. Similar results showing greater K losses from soil types with little ability to retain K have been obtained by Hogg (1968) and Hogg and Toxopeus (1970) in leaching column experiments, and by Muller and McSweeney (1974) in a glasshouse pot trial.

The effect of soil pH on the extent of K leaching has been studied in laboratory experiments. These have shown that raising the soil pH, of soils with pH dependent negative surface charge, can increase K^+ adsorption (Edmeades and Judd, 1980) and therefore decrease K leaching. However, if this pH increase has been associated with the application of Ca (e.g., lime or calcium hydroxide) then the newly created exchange sites tend to be occupied by Ca^{++} and the added Ca^{++} can also displace K^+ from the exchange sites (Edmeades, 1982; Bolan et al., 1988), which may result in an increase in K leaching. Where soil pH is increased in the absence of Ca^{++} by using sodium hydroxide, the increased CEC can increase the amount of K^+ adsorbed from the soil solution (Bolan et al., 1988). Application of dairy cow urine to soils tends to raise soil pH (Doak, 1952). Since the main cations in urine are K^+ and NH_4^+, there may be an increase in K^+ adsorption and reduced leaching losses on soils with pH dependent negative surface charge. No research appears to have been carried out on this effect.

The effect of the rate of K applied on the amount of K leached has been investigated in only a few experiments in New Zealand. While an increased application rate resulted in a greater proportion of the applied K being leached in intact core and pot experiments with a yellow-brown loam soil (Saunders and Metson, 1959; Metson and Saunders, 1962), another pot experiment using a yellow-grey earth soil showed the reverse (Muller and McSweeney, 1977). However, in all of these experiments the amount of K lost via leaching accounted for only a small percentage (<14%) of the K applied.

The type of K fertiliser applied may also influence the amount of K leached. In laboratory leaching column and pot experiments losses of K were shown to be lower when K was applied as a bicarbonate, carbonate or phosphate salt compared with sulphate or chloride salts (Davies et al., 1962; Metson and Saunders, 1962; Hogg and Cooper, 1964). These authors indicated that this effect was caused by the lower solubility of the bicarbonate, carbonate and phosphate forms of K. A second explanation may be that since these anions are also capable of undergoing sorption and precipitation reactions in the soil, there may have been a lower anion concentration in the soil solution and therefore less K was leached. Edmeades and Judd (1980) suggested that on soils dominated by pH dependent charges, addition of carbonate and silicate anions may increase soil pH and therefore the number of soil surface negative charges and cation retention. More recently, coating potassium chloride with sulphur has been shown to be an effective way of reducing K dissolution and leaching in laboratory experiments (Saunders et al., 1988).

2.7.1.2 *Effect of experimental technique on estimates of potassium leaching*

A number of experiments have been carried out to determine the amount of K leached from New Zealand soils, using either laboratory columns of soil (Davies et al., 1962; Hogg, 1968; Hogg and Toxopeus, 1970), pots of sieved repacked soil (Davies and Hogg, 1960; Hogg and Cooper, 1964; Muller and McSweeney, 1974, 1977), intact cores (Saunders and Metson, 1959), lysimeters (Hogg, 1981) or measurements from field drains (Smith et al., 1984). Results from these trials showed a variety in the percentage of applied K that was leached, i.e., 2-22%, <1-61%, 1-15%, 4% and <1% for each different type of experiment respectively. Some of this variation will be due to the chemical factors outlined in section 2.7.1.1, but differences in the techniques used in these experiments will also have a major influence.

Many laboratory experiments have involved leaching a known amount of solute applied as a slug or pulse through a column of sieved soil with moisture contents at or near saturation. Results usually show that if the solute is not adsorbed by the soil its appearance in the leachate will resemble a symmetrical bell shaped breakthrough curve. The peak solute concentration appears after sufficient leachate, equivalent to the total volume of pores in the column (one pore volume), has been collected (Figure 2.4). This type of breakthrough is described as piston-type flow (Bouma, 1981) or miscible displacement (White, 1985). Analysis of the pattern of the breakthrough curves can be used to compare adsorption and ionic exchange between soils and under varying leaching conditions (Nielsen and Biggar, 1961; Cassel et al., 1974; McMahon and Thomas, 1974; Smith and Davis, 1974). Experiments with undisturbed, saturated or unsaturated soil tend to show earlier and steeper breakthrough of the solute (Figure 2.4). This indicates that the solute flow through the soil is bypassing some of the total pore space and moving rapidly down a network of continuous macropores (Elrick and French, 1966; Kissel et al., 1973; Cassel et al., 1974; McMahon and Thomas, 1974; Kanchanasut et al., 1978; Wild and Mazaheri, 1980; Scotter and Kanchanasut, 1981; Tyler and Thomas, 1981; M.S. Smith et al., 1985; Francis, 1986). These macropores include soil cracks, fissures and naturally developed soil pipes, and are also created by soil fauna, plant roots (Beven and Germann, 1982) and mole draining (Bowler, 1980). The breakthrough curve can also be skewed with an extensive tailing off section (Figure 2.4) due to the diffusion of solute into soil aggregates and micropores where displacement by the eluting water is slow (van Genuchten and Wierenga, 1977). This type of breakthrough is referred to as preferential flow in this review, although it has also been described elsewhere as short-circuiting, bypassing, and channeling (Bouma and Dekker, 1978; Beven and Germann, 1982; Smettem et al., 1983).

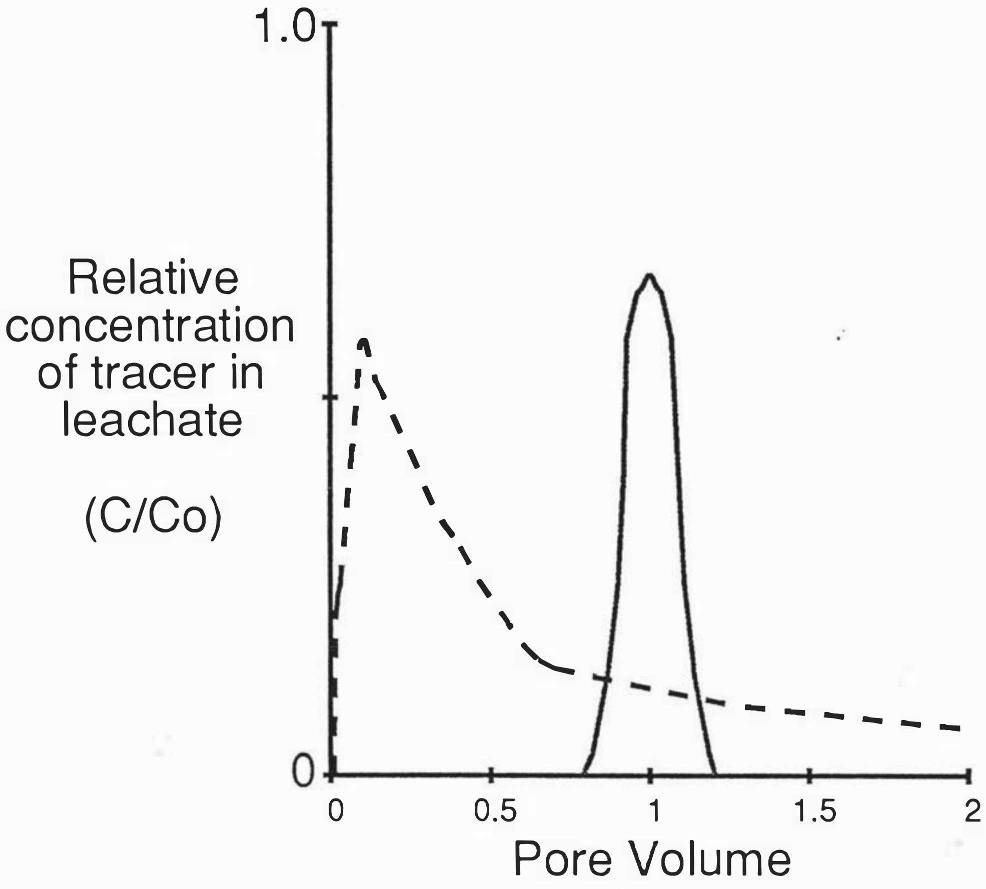


Figure 2.4 Types of breakthrough curve for a pulse of solute moving through a column of saturated soils: piston-type flow (—): preferential flow (---).

Evidence for preferential flow in the literature is numerous and several reviews have been published including those by Thomas and Phillips (1979), Beven and Germann (1982) and White (1985).

In the field, preferential flow will occur under the influence of gravity through macropores when the application rate of water (e.g., rainwater) exceeds the infiltration rate into the soil matrix (Bouma and Dekker, 1978; Bouma, 1981). While preferential flow can result in the rapid loss of solutes that are dissolved or are carried with the water moving through the soil profile (e.g., M.S. Smith et al., 1985), only a small proportion of the soil volume is involved and so the percolating rainwater will bypass solutes that have diffused into soil aggregates (Quisenberry and Phillips, 1976; Thomas and Phillips, 1979; Tyler and Thomas, 1981; White et al., 1986). Therefore the leaching losses of native soil K from undisturbed soil due to preferential flow may not be as high as from piston-type flow in laboratory experiments using sieved soil.

The application rate of dairy cow urine during a urination event is very high (0.2 dm^3 $second^{-1}$; from Goodall, 1951 and Davies et al., 1962) and may initiate preferential flow. The possible occurrence of preferential flow of urine through the soil profile and the significance that this has on K cycling has not been studied.

Very few measurements of K leaching losses have been collected in the field situation under permanent pasture. The literature from field experiments carried out overseas shows that the amount of K leached in subsurface drainage water collected from permanent pastures was small and in the order of <5 kg K ha^{-1} yr^{-1} (Bolton et al., 1970; Burke et al., 1974). However, such trials were conducted in the absence of grazing animals which may increase K losses. In New Zealand, annual losses of 4.4 kg K ha^{-1} have been measured in the subsurface drainage water collected from pasture on an artificially drained yellow-grey earth soil under sheep grazing (Smith et al., 1984), and 14-21 kg K ha^{-1} collected in lysimeters from a northern brown loam under cattle grazing (Steele et al., 1984). Predicted losses from dairy pastures of 13 and 21 kg K ha^{-1} have been made by Campkin (1985) based on measurements of the K concentration in drainage water from the Ruakura Agricultural Research Station No. 5 dairy farm (yellow-brown loam soil) and a peat soil in the Waikato region.

This literature review shows that while research studies on K leaching are numerous, many of the findings relate to laboratory experiments only, and there is little information on the quantity of K that is likely to be leached from grazed pasture.

2.7.2 Loss of Potassium in Hay and Silage Crops

Considerable amounts of K can be lost when pasture is mechanically harvested and removed from the dairy farm in the form of hay and silage crops. For example, a 5,000 kg DM ha^{-1} silage crop with a K concentration of 2.8% represents a loss of 140 kg K ha^{-1}. The loss from the farm is reduced if these crops are fed back to the farm's stock at a later date as there is then some recycling of the K. Conversely, if stock feed is imported on to the farm there is then a gain in plant K (Mott, 1974).

2.7.3 Losses of Animal Potassium

The amount of K that is lost from the cycle due to the dairy animal is the sum of the K in dairy products (milk, culled cows and surplus calves) and the K in excreta which is transferred to unproductive areas of the farm.

2.7.3.1 *Animal products*

The mean concentration of K in the milk of New Zealand dairy cows is 0.13%, ranging from 0.12 to 0.14% over the whole lactation period (September to April; Hutton et al., 1967). Literature from overseas research reports values of 0.12% (Wilkinson and Lowrey, 1973) and 0.15% (Paquay et al., 1969), thus suggesting that the K concentration in milk is relatively constant.

It has been estimated that the amount of K in milk accounts for 5% of the K ingested by cows fed on fresh pasture (Hutton et al., 1967). Overseas research indicates that a higher proportion of the ingested K (13%) may be present in milk (Paquay et al., 1969). However, this value was derived from cows fed a range of concentrates, cereals, straw and silage which had a lower K content than the fresh pasture used in the experiments of Hutton and his co-workers.

The body tissue of cows contains 0.15% K (Middleton and Smith, 1978). Therefore a 400 kg cow that is culled will result in a loss of 0.6 kg K. Similarly selling a 25 kg calf which has a body tissue concentration of 0.17% K (Middleton and Smith, 1978) represents a loss of 0.04 kg K.

2.7.3.2 *Transfer of excretal potassium to unproductive areas*

Unproductive areas of the farm where excreta is deposited include the milking shed, raceways, feeding platforms, and bare ground around gateways, water troughs, camping sites, under trees and next to shelterbelts. On most farms the greatest transfer will be to the milking shed and raceways with only a small amount of excreta being deposited in the other areas. The proportion of excreta that is deposited in the dairy shed and raceways has been estimated at between 2 and 25% by a number of research workers (Table 2.7). Such a variation is presumably due to a variety of management practices between the sites, which affect the length of time the cows spend in the raceway and dairy sheds. There are few measurements in the literature of the time that cows spend off the paddock during milking. The data available indicates a tremendous variation from 2 hours (Hancock, 1950) to 8 hours (Karlovsky, 1975). More data on milking times could be easily gathered from simple farm surveys, and would enable better quantification of the amount of K deposited in unproductive areas.

Although measurements have been made of the proportion of excreta that is deposited in unproductive areas, very few measurements have been made of the actual amount of K deposited there. In New Zealand an estimate of 4 kg K cow^{-1} yr^{-1} deposited in the dairy shed can be made from the data presented by Goold (1980) on the effluent collected from a dairy shed in Northland. A similar calculation shows that the Massey University No. 4 dairy farm has a loss of 6 kg K cow^{-1} yr^{-1} (Clarke and Warburton, 1982). A review of the literature on New Zealand dairy shed effluent by Warburton (1977) gives a range of amounts of K in effluent from 0.07 to 0.16 kg K cow^{-1} day^{-1}, with a mean of 0.11 kg K cow^{-1} day^{-1}. Obviously if farms are suitably equipped to collect and redistribute shed effluent to the pasture then this loss of K is reduced.

No measurements appear to have been made on the amount of excretal K that is deposited in raceways.

2.7.3.3 *Total animal loss of potassium*

The total amount of K that is lost in animal products and transfer, as described in section 2.7.3.1 and 2.7.3.2, can be combined together to give a total loss of animal K which is sometimes referred to as an animal loss factor (ALF; Campkin, 1985). During (1984) estimated that this loss was 15-20 kg K yr^{-1} for a jersey cow, while Middleton and Smith (1978) calculated the loss at 26 kg K cow^{-1} yr^{-1}.

Table 2.7 Percentage of daily urinations and defaecations that are deposited in the milking shed and raceways by dairy cows.

Reference	Site	Urinations (%)	Defaecations (%)
Castle et al. (1950)	Reading, UK.	6	12
Hancock (1950)	Ruakura	3	2
Goodall (1951)	Palmerston Nth	11	11
Hardison et al. (1956)	Virginia, USA.	19.7	23.8
Wardrop (1963)	Somerset, UK.	23.5	24.2
Cooper (cited in Steele 1982)	Ruakura		12
During (1984)	NZ.	10-15	

Campkin (1985) has proposed an ALF of 4 kg K per stock unit (su) for a flat dairy farm and 4.5 kg K su^{-1} for a dairy farm on rolling hill slopes. No justification is given for the higher loss on the rolling country but presumably this is to allow for some transfer of excretal K to campsites using similar ideas to those developed from research with sheep conducted by Gillingham and During (1973). Basing the loss on a stock unit rather than per cow allows for differences in pasture intake that occur between cows of different liveweight and productivity. Thus a 300 kg cow producing 120 kg milkfat yr^{-1} equals 5.4 su, while a 500 kg cow producing 180 kg yr^{-1} milkfat equals 7.8 su (Cornforth and Sinclair, 1984). Using Campkin's (1985) ALF values the amount of K lost by these two cows is 20 kg K cow^{-1} and 35 kg K cow^{-1} respectively.

The difference between the K loss values proposed by Middleton and Smith (1978), During (1984) and Campkin (1985) are mainly due to a variation in the proportion of excretal K that was considered to have been transferred to unproductive areas. During (1984) used a value of 10-15%, while Middleton and Smith (1978) based their calculations on a 25% loss derived from Karlovsky (1975), and Campkin (1985) used a loss of 18%. Such losses tend to be at the high end of the range presented in Table 2.7, indicating that on some farms the amount of K lost may be less than the 15-35 kg K cow^{-1} proposed by these authors. Clearly since the losses of animal K can represent a major loss of K from the soil-plant-animal cycle, this aspect needs to be researched further.

2.7.4 Conclusions

One pathway for losses of K from the soil-plant-animal cycle of a pasture system is via leaching. Most studies on leaching losses have been confined to laboratory and glasshouse studies. Whether these results apply to the field situation has not been clearly established. The few field measurements of annual K losses that relate to dairy farming on New Zealand soils include those reported by Campkin (1985) and the trial results of Saunders and Metson (1959). Data from these two sources is limited to two soil types and indicate that up to 20 kg K ha^{-1} may be leached annually. Such a loss represents a considerable proportion of the 60 to 70 kg K ha^{-1} proposed by During (1984) as an annual fertiliser application for dairy farms. The amount of K lost from other soil types is unknown. Also unknown is whether preferential movement of dairy cow urine occurs down through the soil profile and what effect this has on the losses of K or on the recycling of urine K.

While it is possible to quantify the losses of K that occur via hay, silage, and dairy products, there appears to be some confusion over the proportion of excretal K that is transferred out of the cycle to unproductive areas, as values in the literature range from 2 to 25%.

2.8 MODELLING OF POTASSIUM CYCLING

The information provided by this literature review has enabled quantification of the forms and amounts of K involved in the K cycle in dairy pastures. Some of the important mechanisms that influence the fate of fertiliser K have been identified. With this information a mass balance approach can be used to model the redistribution of K within a farming system. From such a model the probable K losses can be calculated and therefore the amount of fertiliser required by the system can also be calculated.

In this section of the literature review some examples of the models that have been developed are examined with particular emphasis on the CFAS model, which was developed for New Zealand pastures by New Zealand MAF scientists. Finally, information from the earlier review is used to identify areas where more data should be obtained to improve the CFAS model.

Several models have been published for K cycling in grazed pastures (Wilkinson and Lowrey, 1973; Adams and McAllister, 1975; Newbould and Floate, 1977; Middleton and Smith, 1978; Till, 1981; Campkin, 1985). Basically, these models are balance sheets of inputs and outputs of the K cycle, and the difference between outputs and inputs is used as an estimate of fertiliser requirements (Adams and McAllister, 1975; Middleton and Smith, 1978; Campkin, 1985).

The simplest model is that of Adams and McAllister (1975) which calculates the difference between the amount of K produced in excreta of housed livestock and the amount of K needed to grow the crops that feed these stock. However, this model does not account for losses of K due to leaching or for changes in the plant available soil K^+ pool due to the supply of K from nonexchangeable sources and the weathering of soil minerals.

The other models (Wilkinson and Lowrey, 1973; Middleton and Smith, 1978; Campkin, 1985) are more complex as they involve animals grazing on pasture rather than housed indoors, this means that there is less control on the amount of K ingested by the animals and where the excreta is deposited. Of these models only those of Wilkinson and Lowrey (1973) and Campkin (1985) account for the contribution of soil nonexchangeable K^+ as sources of plant available K^+. From the review of literature it has been shown that on many New Zealand soils nonexchangeable K^+ is an important source of plant K^+ (providing up to 200 kg K ha^{-1} yr^{-1}; section 2.2.2) and so should be included as part of the a model on the K cycle.

A model which predicts fertiliser K for any individual farm in New Zealand has been developed by Campkin (1985). A slightly modified version of this is used as the MAF's CFAS model for predicting K fertiliser requirements (Cornforth and Sinclair, 1984). The CFAS model is a "depletive" model rather than a true "maintenance" model because a supply of exchangeable and nonexchangeable soil K^+ is used to offset the losses of K that occur. Ultimately, this supply will deplete the amount of K in the soil. The CFAS model can be modified easily for all forms of stock farming, but this review will relate to dairy farming only.

The model comprises three main parameters:

1.animal losses [SRxALF]

2.leaching losses [(R+I-PE)x((SR/1250)xleaching factor)]

3.contribution of soil K [10(QTK-Kc)]

From these parameters and the following formula the maintenance K requirement is calculated:-

$$K\ (\mathrm{kg\ ha^{-1}}) = [SR \times ALF] + [(R+I-PE) \times ((SR/1250) \times \text{leaching factor})] - [10(QTK-Kc)]$$

1.<u>Animal loss</u>. The amount of K lost from the farm is based on the stocking rate (SR) and an ALF. The SR is the number of stock units ha^{-1} where one stock unit has an intake of 550 kg DM yr^{-1} (section 2.7.3.3). The ALF represents the amount of K ingested by each stock unit that is lost in animal products (milk, calf and culled cow) and in transfer to the raceway and dairy shed when the cows are milked (section 2.7.3.3).

2.<u>Leaching loss</u>. Firstly the annual drainage is calculated from the mean annual rainfall (R) plus any irrigation applied (I) minus the mean potential evapotranspiration (PE). Assuming that this drainage water has a mean K concentration of 2 μg ml^{-1}, the amount of K that is lost due to leaching is calculated and converted to kg K ha^{-1} with the constant 1250. The K loss is then adjusted for the number of stock (SR) and soil type (leaching factor). An empirical leaching factor of 1 is used for most soil groups except for peat, yellow-brown pumice and yellow-brown sand soils which have a leaching factor of 2. The leaching factor of 2 is to allow for the higher K concentration (4 $\mu g\ ml^{-1}$) in the drainage water of these soil types. Since the introduction of the model the leaching factor for yellow-brown loam soils has been increased from 1 to 2 based on the recommendation of MAF staff who were using the CFAS model to predict K fertiliser requirements on this soil type (I.S. Cornforth, pers. comm.).

3.Soil supply. The amount of K supplied by the soil is estimated empirically from the difference between long term K supplying power of the soil (based on the Kc soil test) and short term K supply (based on QTK). For soils with nonexchangeable sources of plant available K^+ (e.g., recent soils) the TBK test replaces the QTK. These three tests have been described previously in section 2.6.1.4.

Additional modifications can be made if hay and silage crops are removed from the farm.

The parameter in the model which has the greatest effect on fertiliser requirements is the measurement of K supply by the soil. Unfortunately, the 30% variation (I.S. Cornforth, pers. comm.) associated with the QTK makes this soil factor extremely variable and this has a dramatic effect on fertiliser recommendations. Thus for a site with a true QTK value of 11, the range of measured QTK values is between 8 and 14. Therefore the fertiliser requirement for this site may vary from 20 to 80 kg K ha^{-1} depending on the value of the measured QTK. From a cost point of view this may affect fertiliser expenditure by up to \$30 ha^{-1} (1988 ex works price of K = 50c kg^{-1}).

Apart from the large variability that may occur in the QTK, the main limitation to the model is the lack of suitable data with which to confirm the accuracy of its predictions. Campkin (1985) regressed observed maintenance K fertiliser rates against predicted K fertiliser rates for eight dairy farms and the mean result from seven experiments under dairy grazing (O'Connor and Feyter, 1980) and obtained a good correlation (r=0.76). As no information is provided on how the observed maintenance rates were derived for the eight farms, it is presumed that they were rates which were maintaining soil K values at an adequate level for plant growth as measured by the QTK. All of these farms were on yellow-brown loams in the Waikato and had medium to high QTK values for K. Some of these soils are thought to have a source of nonexchangeable K^+, which would have been capable of replenishing the exchangeable K^+ pool as measured by the QTK (Hogg, 1968). Thus the observed fertiliser rates may appear to be maintaining QTK even if the rate was too low to do this adequately.

Further testing of the accuracy of the model's predictions is currently being carried out by the MAF. This testing is in the form of a monitoring programme aimed at determining the relationship between K fertiliser recommendations and the K status of soil and pasture samples (I.S. Cornforth, pers. comm.). The monitoring programme involves collection and analysis of soil and plant samples twice a year from 150 sites throughout New Zealand on a range of soil types. Detailed records of fertiliser applications are also kept. So far data has been collected for 4 years and a preliminary interpretation of the results has begun.

Revisions to the CFAS model are being planned currently by MAF staff. These revisions are based on the experience of researchers and advisers who are involved in making predictions of fertiliser requirements to farmers. Since the model was developed there has been little in the way of new research data which can be used to update and refine the model. Proposed revisions are aimed at making the model more of a maintenance model by introducing the concept of a maintenance soil K value as measured by the QTK (subsequently referred to as QTKm) (A. Metherell, pers. comm.). For an individual farm the QTKm value can be estimated as that soil QTK level which from experience is required to maintain a certain stocking rate on a particular soil type. The fertiliser recommendation will then be calculated using the same animal loss and leaching loss parameters of the original model, and a standard value for the amount of K supplied by each soil type (e.g., 10, 30 and 50 kg K ha^{-1} for soils with low, medium and high Kc status respectively). The fertiliser recommendation can then be modified according to the measured QTK, i.e., if the measured QTK is less than QTKm then more fertiliser is applied; conversely less fertiliser is applied if the measured QTK is greater than the QTKm.

At present the usefulness of the CFAS model's predictions of fertiliser requirements is limited by the highly variable estimates of the amount of plant available K^+ in the soil and by a poor quantification of K losses. This thesis addresses the latter topic as the current revisions by MAF scientists are concerned with the supply of soil K. The need for further research on the losses of K from grazed pasture is justified on the basis that in the reviewed literature there is a conflict over the value of some of the parameters used. For example, the ALF is based on 18% of the ingested K being transferred to unproductive areas (Campkin, 1985), while the data in Table 2.7 suggest that this loss may be too high. Similarly the amount of K leached from the grazed pasture is based on estimates of the K content in the drainage water of 4 $\mu g\ ml^{-1}$ for peat, yellow-brown loam, yellow-brown pumice and yellow-brown sand soils and 2 $\mu g\ ml^{-1}$ for all other soils. However, these values have been derived from a mixture of field observations and laboratory experiments (Campkin, 1985). As discussed in section 2.7.1.2 extrapolation of the laboratory results to the field situation is subject to limitations.

2.9 CONCLUSIONS

It is apparent from this literature review that as K cycles through grazed dairy pasture major losses can occur, particularly from the animal K compartment (Middleton and Smith, 1978; Campkin, 1985). If these are not balanced by inputs of fertiliser the soil K levels will decline.

There is sufficient data available to construct a model for K cycling in a grazed dairy pasture in the form of a balance sheet of losses and gains. Such a model is used by the New Zealand MAF to predict fertiliser requirements. The usefulness of the MAF's CFAS K model is limited by a lack of testing of the accuracy of the animal loss and leaching loss parameters in the field, and its reliance on measurements of soil exchangeable K^+ (which tend to be extremely variable in grazed pastures).

The available data on the amount of K that is lost through transfer by animals and leaching, and on the fate of animal K that is returned to the soil pool in urine, is extremely variable and difficult to interpret due to the wide range of conditions under which the data was collected. For example, data on the proportion of urine K that is leached from urine affected soil shows a range of between 1 and 72% (Table 2.6). The variation may be due to a number of factors including soil type, the type of experiment and the type of leaching events that were applied (section 2.6.4.1). Obviously a more accurate prediction of K losses requires a better understanding of the factors involved than is given by this data. Better quantification of leaching losses and the fate of urine K will also require a definition of the depth of soil from which plant roots extract the majority of their K requirement. Research into these aspects will allow more accurate predictions of K fertiliser requirements for grazed dairy pastures.

CHAPTER 3

PLANT UPTAKE OF POTASSIUM FROM DIFFERENT DEPTHS OF URINE AFFECTED SOIL

3.1 INTRODUCTION

Of the K ingested by dairy cows (315 kg K ha^{-1}; section 2.4), approximately 80% is excreted in urine (section 2.4). If 10% of the excreta is deposited in areas other than pasture (e.g., dairy shed and raceways; section 2.7.3.2), then the remaining 72% of the ingested K (approximately 230 kg K ha^{-1} yr^{-1}) is voided on to the pasture in urine spots. The amount of K deposited in one urine spot (average surface area = 0.19 m^2; section 2.6.3.4) is equivalent to an application of 1000 kg K ha^{-1} (Saunders, 1984), which is excessive compared with the 60-70 kg K ha^{-1} that is usually applied in fertiliser (During, 1984). Therefore, the ability of pasture plants to reutilise the K from urine affected soil will have a major influence on the amount of fertiliser K required by the dairy farming system.

Plant uptake of urine K from the soil is dependent on a number of factors, including the distribution of active roots and urine K through the soil profile, the soil moisture profile and the presence of urine N (Barber, 1984; section 2.6.1). The plant roots of most pasture plants are found in the surface layers of the soil profile (Williams, 1968; Caradus and Evans, 1977) and it has been shown by Australian (Ozanne et al., 1965) and American (Peterson and Smith, 1973) research workers that the majority of K taken up by pasture plant species is derived from the surface layers of soil. In New Zealand, measurements on the uptake of phosphorus (P) and sulphur (S) by pasture plants have also shown that the majority of root absorption occurs close to the soil surface (Jackman and Mouat, 1972; Gregg et al., 1977; Gillingham et al., 1980; Horne, 1985). No direct measurements have been made on the proportion of plant K taken up from different depths of New Zealand pasture soils. Over a period of 7.5 years, Weeda (1978) measured a decrease in the amount of soil exchangeable K$^+$ which occurred throughout the top 30 cm of a pasture soil. These decreases were interpreted to be the result of plant uptake. During and Campkin (1980) monitored the uptake of K by ryegrass plants over a 5 year period at 10 sites throughout the northern North Island. At 3 of these sites, where the amount of K recovered by the herbage could not be accounted for by changes in exchangeable K$^+$ in the 0-25 cm depth of soil, they concluded that ryegrass was capable of extracting K from soil depths greater than 25 cm. At 2 sites they analysed soil samples from deeper down the soil profile. A decrease in exchangeable K$^+$ over time was noticed, which suggested that ryegrass plants were able to extract K down to 61 cm. Apart from this research, no other studies describing the soil depth from which the majority of plant K is derived in New Zealand pastures have been published.

Most New Zealand soils have naturally high K contents (Table 2.1) and although the plant available K pool is only a fraction (1-5%; Table 2.1) of the total K content, there is a continual release of nonexchangeable and structural K into the available pool. Therefore it can be difficult to quantify the uptake of K from any one soil depth by measuring decreases in the plant available K pool directly. An alternative method of investigating plant uptake of K is to place readily distinguishable analogues (tracers) of K at known soil depths, and then study their exchange with the plant available K pool and their uptake by plants. Radioactive isotopes of K are suitable analogues but have received little use, mainly because ^{42}K has a short half life (12.4 hours) and ^{40}K has a low activity and is expensive. These restrictions have led to the use of rubidium (Rb) as an analogue for K by many researchers. Although there are some differences in behaviour between these elements, particularly during soil fixation (Fried et al., 1959; Oien et al., 1959), Rb has been used successfully as an analogue for relative uptake of K from different depths of the soil profile in a pot experiment (Walker and Barber, 1962).

To gain valid estimates of plant uptake of nutrients in field studies it is essential to label the soil with a tracer with minimal disturbance of the soil structure. Previous attempts with K and other nutrients have involved applying the tracer via tubes inserted horizontally into the soil from pits dug alongside the plots (e.g., Ozanne et al. (1965) for K and Goh et al. (1977) for S). Other experimenters have injected the tracer into the soil via a hypodermic syringe inserted in to the soil to a specified depth (e.g., Newbould et al., 1971). Recently an injector apparatus has been developed at Massey University which allows horizons of the soil profile to be labelled vertically *in situ* with minimal disturbance to the soil. This injection apparatus was used (in this experiment) to label chosen depths of the soil profile with Rb.

3.1.1 **Objectives**

The overall objectives of this field experiment were to:

i. determine the amounts of herbage K derived from various depths of urine affected soil.

ii. determine the importance of the plant uptake of urine K as a pathway for K conservation in the K cycle of dairy pastures.

In order to achieve these objectives Rb was used as an analogue for K. The objectives of using Rb were to:

i. label soil depths uniformly with Rb as an analogue of K.

ii. measure plant uptake of Rb from those depths expressed as a function of the total exchangeable pool of Rb at each depth.

iii. calculate relative root activity for Rb uptake and therefore K uptake.

3.2 MATERIALS AND METHODS

3.2.1 Preliminary Investigations Into the Measurement of Rubidium In Soil and Plant Samples

The Rb^+ concentration (range 0.2-1 μg ml^{-1}) in soil and plant extracts can be measured by atomic emission in the presence of 0.2% caesium chloride and 0.2% potassium chloride to suppress ionisation of the Rb^+ ions (Sanui and Pace, 1968). Preliminary analysis of plant and soil extracts by this method showed that the concentration of Rb^+ was influenced by the concentration of K^+ in the plant and soil samples. Measuring the Rb^+ concentration in a series of samples with increasing K^+ concentration showed an increase in Rb^+ regardless of whether there was any Rb^+ in the samples or not (Table 3.1a). To overcome this problem the plant and soil extracts were diluted until K did not interfere (the concentration of caesium chloride and potassium chloride in the extracts remained at 0.2%). For these samples the Rb^+ concentration was diluted to 0.01-0.05 μg ml^{-1}. This method appeared to minimise the interference from the background K (Table 3.1b) but meant that the large dilutions required before Rb could be measured resulted in a multiplication of experimental error.

The efficiency with which Rb could be extracted from soil was tested in the laboratory by mixing rubidium chloride with 3 replicate units of dry soil (4 g of Tokomaru silt loam plus 1 ml of 1.17 x 10^{-4}M Rb). After a short period of time, the Rb was extracted with 1M ammonium acetate as described in section 3.2.5.1. It was possible to recover 104% (SEM = 2.3) of the Rb by this technique. It was probable that the quantity of background soil exchangeable K^+ was responsible for the slight over estimate of Rb recovery.

Table 3.1a The measured concentration of Rb ($\mu g\ ml^{-1}$) in herbage digests and 1M ammonium acetate soil extracts containing various K concentrations. The herbage digests and soil extracts were spiked with either 0 or 1 $\mu g\ ml^{-1}$ Rb. Values are means of duplicate experimental units (mean deviation <8%).

K^+ concentration ($\mu g\ ml^{-1}$)	Amount of Rb added ($\mu g\ ml^{-1}$) 0	1	Δ Rb
Herbage digests			
0	0.0	1.0	1.0
90	0.08	0.54	0.46
210	0.12	0.53	0.41
390	0.24	0.65	0.41
490	0.56	0.86	0.30
1900	1.66	1.76	0.10
Soil extracts			
0	0.0	1.0	1.0
10	<0.01	0.69	0.69
20	0.19	0.84	0.65
70	0.42	0.82	0.40
120	0.65	0.94	0.29

Table 3.1b As for Table 3.1a, but all digests and extracts were diluted twenty times. Further details are given in the text (section 3.2.1).

K^+ concentration ($\mu g\ ml^{-1}$)	Amount of Rb added ($\mu g\ ml^{-1}$) 0	1	Δ Rb
Herbage digests			
0	0.0	0.05	0.05
4.5	0.00	0.04	0.04
19.5	0.00	0.06	0.06
95	0.00	0.05	0.05
Soil extracts			
0	0.0	0.05	0.05
0.5	0.00	0.05	0.05
1	0.00	0.06	0.06
3.5	0.00	0.04	0.04
6	0.00	0.05	0.05

3.2.2 **Field Experiment**

3.2.2.1 *Trial site*

The trial site was on established pasture on the Massey University No. 4 dairy farm, 5 km east of Palmerston North. The pasture comprised mainly perennial ryegrass and white clover. The soil was a Tokomaru silt loam and some of its properties are given in Table 3.2. The site had been artificially drained with tile and mole drains placed at depths of approximately 75 cm and 45 cm respectively.

In June 1987 an area (5 m x 3 m) was fenced off from stock. The pasture was mown to a height of 2 cm above ground level and 21 cylinders (15 cm in diameter and depth) were hammered into the soil to isolate each soil "core". A fortnight later (8 July 1987) the herbage on each soil core was trimmed to 2 cm and the treatments were applied.

3.2.2.2 *Urine and rubidium chloride applications*

Two hundred ml of dairy cow urine were applied to each core to simulate a urination event of 11 $dm^3\ m^{-2}$ (Hogg, 1968; 1981). The urine was collected from the Massey University No. 3 dairy farm during a spring milking, frozen immediately and stored until required. The concentration of K^+ in the urine was 0.66%.

Twenty to 60 minutes after the urine had been applied, a solution of rubidium chloride (0.286 mg Rb ml^{-1}) was injected into the cores at three different depths (0-2.5, 0-14.5 and 12-14.5 cm) with seven replicates of each. The injection apparatus comprised twenty 18 cm long stainless steel tubes (16 gauge) spaced evenly on a 15 cm diameter stainless steel disc. Prior to insertion of the stainless steel tubes into the soil a template with stainless steel pins, of a similar diameter to the tubes, was pushed into the soil core to the required depth. As the stainless steel tubes were withdrawn slowly from the soil, rubidium chloride was injected from a port on the side of each tube into the soil at the desired soil depths. The volume of the solution and the amount of Rb injected in each treatment is given in Table 3.3. Only a small volume of solution (<1% of the total soil water content) and only a small amount of Rb (equivalent to <0.5% of the amount of exchangeable K^+) was applied at each depth. Therefore, the soil moisture content and the soil exchangeable K^+ pool at each soil depth remained essentially unaltered.

Table 3.2 Background information on the Tokomaru silt loam.

N.Z. soil group	yellow-grey earth
Soil taxonomic classification	Typic Fragiaqualf
pH[1]	6.6
Cation exchange capacity[2] (me $100g^{-1}$)	22
Organic carbon[3] (%)	5.2
Dominant clay minerals[4]	illite vermiculite

[1] in water; section 3.2.5.2
[2] section 3.2.5.3
[3] Tabatabai and Bremner (1970)
[4] Pollok (1975)

Table 3.3 The volume of solution and amount of Rb injected into the soil for each treatment.

Depth of soil labelled (cm)	Volume (ml)	Rb (mg)
0-2.5	4.0	1.14
0-14.5	21.0	6.01
12-14.5	3.6	1.03

After all the treatments had been applied, three cores (one for each Rb placement depth) were dug out and taken back to the laboratory for determination of the amount of exchangeable K^+ and Rb^+ in each soil depth (section 3.2.5.1). This provided measurements for the accuracy of the placement of Rb in the soil.

3.2.2.3 *Herbage and soil sampling*

At approximately five weekly intervals, the herbage on all the cores was harvested, dried at 65°C for 24 hours, and weighed. The dried herbage was then ground finely and the K^+ and Rb^+ concentrations in the herbage were measured using the method described in section 3.2.4. After the herbage had been harvested, three cores (one per Rb placement depth) were retrieved, removed to the laboratory and the K^+ and Rb^+ concentrations in each soil depth were measured (section 3.2.5.1). This provided information on the K and Rb content of the soil for most of the sampling dates (at the February sampling only the herbage was collected). Soil samples from beneath each core (14.5-18 cm depth) were also collected and the K^+ and Rb^+ concentrations measured to determine if there was any leaching of Rb beyond the 14.5 cm depth.

Meteorological data were obtained from the nearest meteorological station 1 km away.

3.2.2.4 *Measurement of the distribution of pasture root mass with depth*

The distribution of pasture root mass with soil depth on the Tokomaru silt loam was measured adjacent to the field trial area. At ten sites selected at random throughout the paddock a pit was dug in the soil to a depth of 30 cm. Soil samples of a known volume (491 cm^3) were collected at 5 cm depths down to 30 cm. The samples were washed with water to separate the roots from the soil, then the roots were dried at 65°C for 24 hours and weighed. The samples were collected from the field in October (1985).

3.2.3 **Glasshouse Experiment**

Some aspects of the field experiment were repeated in a glasshouse experiment which was designed to measure the amount of K and Rb that may have been leached from the soil cores during periods of heavy rain. Of particular interest was whether or not any leaching resulted from the heavy rain storms that fell on the field cores during the 14th week of the trial (between the 8th and 13th of October 1987).

Five intact soil cores of the same dimensions as used in the field experiment were collected from the pasture trial area. The herbage was trimmed to 2 cm above the soil surface and a piece of nylon gauze was secured with a rubber band to the bottom of the core. The cores were placed in trays of water for 10 days, after which time they were allowed to drain for 24 hours. The weight of the cores after drainage was defined as "core capacity" and their soil moisture content (mean gravimetric water content 0.40 g g^{-1}, SEM = 0.03) was considered to resemble "field capacity" in the field.

As in the field experiment, 200 ml of dairy cow urine were applied to each core, using the same batch of urine. The leachate was collected, weighed and the K^+ concentration measured by atomic emission in the presence of 0.2% caesium chloride. Three of the cores were injected at 0-14.5 cm depth with rubidium chloride (5.7 mg Rb per core) using the same apparatus and technique used in the field. No evidence was found that the Rb moved out of the bottom of the core following injection.

In the remaining 2 cores the template of stainless steel pins was pushed into the core but no rubidium chloride was injected. The purpose of these 2 cores was to provide samples of leachate which contained K but no Rb. These samples were used as K background controls for measuring Rb in the leachate from the treated cores.

All soil cores were placed in the glasshouse and watered daily to core capacity by applying distilled water from a wash bottle. The daily watering events did not initiate leaching. After one month the heavy rain storm events which fell on the field trial in October 1987 were simulated. These involved a period of 6 consecutive days of 14, 20, 5, 0, 5 and 23 mm of rain. On each rainy day the simulated rain was applied in events of 5 mm, with a time interval of 2 hours between. Each rain event comprised 90 ml (except for the last event of the first and last rain days, which comprised 72 and 54 ml respectively) applied from a wash bottle. The weight and the K^+ and Rb^+ concentration of the leachate from each rain event were determined as previously described.

At the conclusion of this period of rain events, the exchangeable Rb^+ concentration in each soil core was determined as described in section 3.2.5.1.

An application of 200 ml of 0.1% methylene blue dye (tetramethylthionine chloride) was made to one of the cores that had not been injected with Rb, to identify pathways that the urine and simulated rainfall may have taken through the soil core. Of particular interest was whether the rainfall would be directed down the channels created by the injection apparatus.

Observations of the dye pathways through the soil core showed that this was not the case. Instead the dye moved through a series of macropores created by root channels, earthworms and soil fracture planes. It appears that in a well structured soil such as the Tokomaru silt loam, the channels created by the injection apparatus, some of which were still apparent one month after injection, had very little effect in directing the movement of water through the soil core.

3.2.4 **Herbage Analysis**

Samples of dried ground herbage (0.1 g) were digested with 4 ml of nitric acid at 150^{o}C for 4-5 hours until the extract was clear. The excess liquid was evaporated off at 200^{o}C and the residue dissolved in 5 ml of 2M hydrochloric acid. The K^+ concentration in this solution was determined by atomic emission in the presence of 0.2% caesium chloride, and the Rb^+ concentration was determined as outlined in section 3.2.1.

3.2.5 **Soil Analysis**

3.2.5.1 *Exchangeable potassium and rubidium*

The soil cores were dried in the oven at 65^{o}C for 1-3 days depending on their moisture content, after which time it was possible to remove the cylinder from around the soil core. The cores were then sliced into 3 depths (0-2.5, 2.5-12 and 12-14.5 cm) which were left to dry at room temperature. After sieving (<2 mm; any root material that did not pass through the sieve was finely ground and added to the soil) the samples were extracted with ammonium acetate solution to remove exchangeable K^+ and Rb^+ (Thomas, 1982). Four g of soil were shaken with 20 ml of 1M ammonium acetate in 50 ml polypropylene, screw-capped centrifuge tubes on an end-over-end shaker for 30 minutes. After shaking the tubes were centrifuged (10,000 rpm for 5 minutes using a Sorvall RC5C centrifuge) and the supernatant collected after filtering through Whatman No. 5 filter paper. This process was repeated and the two extracts combined together, prior to adjusting the volume to 50 ml. The K^+ concentration in each extract was measured by atomic emission in the presence of 0.2% caesium chloride and the Rb^+ concentration was determined as outlined in section 3.2.1.

3.2.5.2 *Soil pH*

Ten g of sieved, dried soil were mixed with 25 ml of distilled water and allowed to stand overnight before the pH of the suspension was determined using a combination glass electrode and a Radiometer PHM82 standard pH meter.

3.2.5.3 *Soil cation exchange capacity*

The cation exchange capacity of the soil was measured using a method of Hesse (1977). Five g of dried sieved soil were shaken with 25 ml of 1M ammonium acetate in 50 ml polypropylene, screw-capped centrifuge tubes for 30 minutes on an end-over-end shaker. The tubes were centrifuged (10,000 rpm for 5 minutes), the supernatants carefully decanted and the process repeated. The samples were then washed 3 times with 95% ethanol to remove excess NH_4^+ ions in the soil solution. For each wash 10 ml of ethanol were added; the tubes were then shaken, centrifuged and the supernatant decanted. The NH_4^+ ions adsorbed by the soil were extracted by adding 25 ml of 0.25M barium chloride to the tubes and shaking the tubes for 10 minutes. The supernatant was recovered after centrifugation and filtration. The extraction was repeated and the two extracts were combined prior to adjusting the volume to 100 ml. A 25 ml subsample of the 0.25M barium chloride was taken, 2 drops of phenolphthalein indicator were added with just enough 0.02M sodium hydroxide to turn the indicator pink. Five ml of 40% neutralised formalin were then added and after 5 minutes the solution was titrated with 0.02M sodium hydroxide to neutralise the sample. The cation exchange capacity of the soil was calculated from the number of moles of sodium hydroxide required to neutralise the H^+ ions generated in the reaction between formalin and the NH_4^+ ions as shown by the following equation:-

$$4NH_4^+ + 6CH_2O \rightleftharpoons C_6H_{12}N_4 + 4H^+ + 6H_2O$$

3.3 RESULTS AND DISCUSSION

3.3.1 Fate of Rubidium Injected Into the Soil

3.3.1.1 *Distribution in the soil*

The amounts of exchangeable Rb^+ measured in sections of the soil core immediately after the rubidium chloride was injected into the soil cores showed that the Rb had been placed accurately at the various soil depths (Figure 3.1). However, there was an uneven distribution of Rb with depth in the 0-14.5 cm treatment. In the cores analysed immediately after injection (8 July), 101-111% of the applied Rb was recovered from the 0-2.5 and 0-14.5 cm treatments (Table 3.4). Only 60% was recovered from the 12-14.5 cm treatment (Table 3.4). This lower recovery may have been due to "loss" of Rb below the 14.5 cm depth due either to injection of Rb below the required depth due to surface irregularities, or to the injection of Rb into a macropore through which it flowed beyond the 14.5 cm depth. Unfortunately no samples were

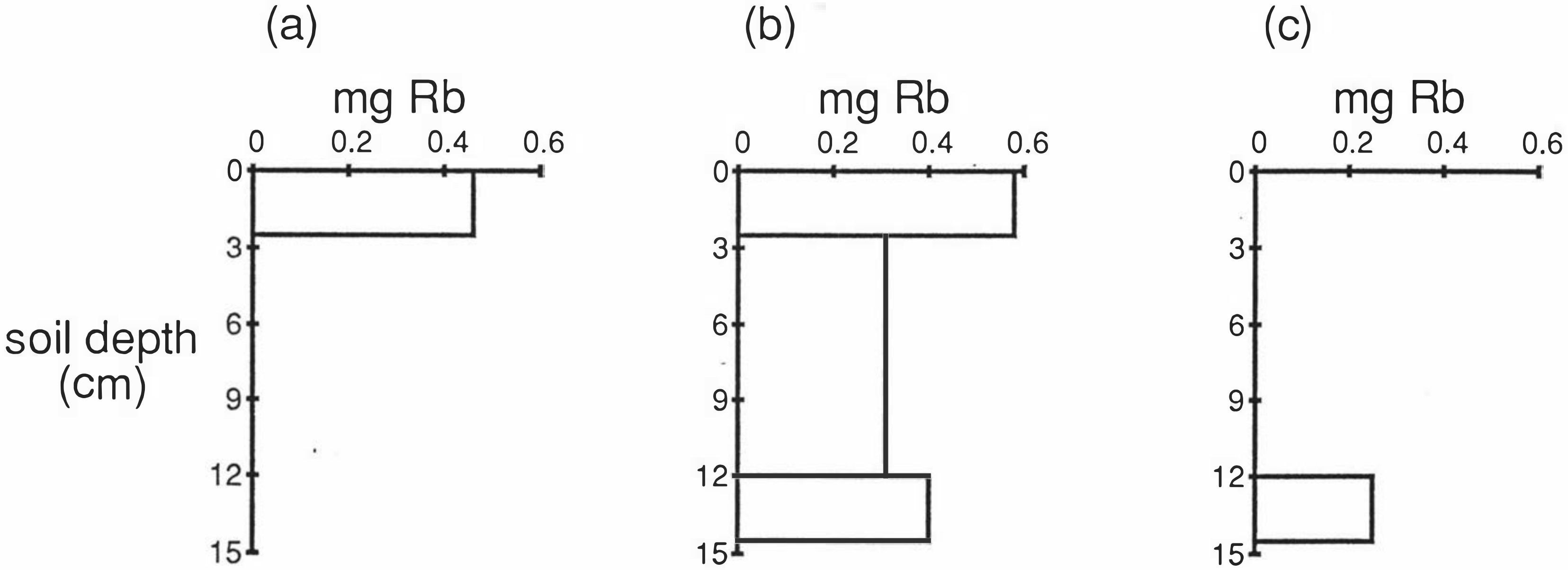


Figure 3.1 The distribution of exchangeable Rb^+ in soil cores immediately after the injection of 3.3×10^{-3}M RbCl at the (a) 0-2.5, (b) 0-14.5 and (c) 12-14.5 cm soil depths.

Table 3.4 The amounts of exchangeable Rb^+ extracted from soil at various harvest dates after the injection of 3.3×10^{-3}M RbCl at 0-2.5, 0-14.5 and 12-14.5 cm soil depths. Values are of one replicate only.

Soil depth Injected		Amount (mg Rb $core^{-1}$) of Rb extracted from each depth of soil (cm)			% of added Rb
Harvest date	0-2.5	2.5-12	12-14.5	14.5-18	recovered
0-2.5 cm treatment					
8 Jul 1987	1.26	0	0		111
11 Aug	1.20	0	0	0	105
16 Sep	0.83	0	0	0	73
19 Oct	0.14	0	0	0	12
26 Nov	0	0	0	0	0
4 Jan 1988	0.07	0	0	0	6
11 Mar	0.18	0	0	0	16
0-14.5 cm treatment					
8 Jul 1987	1.60	2.89	0.99		101
11 Aug	0.66	1.76	0.17	0	43
16 Sep	0.75	3.57	0.89	0	90
19 Oct	0.17	0	0	0	3
26 Nov	0	0	0.57	0.16	12
4 Jan 1988	0	0	0.20	0.22	7
11 Mar	0.20	1.17	0.71	0.08	36
12-14.5 cm treatment					
8 Jul 1987	0	0	0.62		60
11 Aug	0	0	1.02	0	98
16 Sep	0	0	1.26	0	121
19 Oct	0	0	0	0	0
26 Nov	0	0	0	0	0
4 Jan 1988	0	0	0.13	0.26	38
11 Mar	0	0	0.41	0.12	51

collected at this time from the 14.5-18 cm depth to confirm whether the missing Rb was further down the soil profile. At the next two sampling dates (11 August and 16 September) 98-121% of the Rb was recovered from the 12-14.5 cm depth treatments (Table 3.4), therefore it seems that the reason for the low recovery at the first soil sampling was poor injection technique in that soil core. At dates after the 16th of September, the recovery of exchangeable Rb^+ from the majority of soil cores decreased markedly. In three cores no Rb could be detected (Table 3.4). This apparent sudden loss of exchangeable Rb^+ remains unexplained but it is also reflected in the pattern of Rb uptake by the herbage (Table 3.5).

3.3.1.2 *Plant uptake*

The amount of Rb taken up by the herbage was extremely variable between harvest dates and between replicates (Table 3.5). Most of the variation between replicates was due to differences in Rb^+ concentration rather than herbage yield, as the variation in yields tended to be small at each sampling date (Figure 3.2).

The pasture plants absorbed Rb from the 0-2.5 and 0-14.5 cm depths of soil but no Rb (apart from 4.3 μg Rb in one core at the first harvest; Table 3.5) was taken up from the 12-14.5 cm depth. This pattern of Rb uptake coincided with the root distribution in the soil profile. The distribution of the weight of plant roots with depth shows that within the top 15 cm of the soil profile the majority of the plant roots were found in the 0-5 cm (53%) and 5-10 cm (23%) depths (Table 3.6).

Virtually no Rb was taken up by the herbage after the 19-10-87 harvest (Table 3.5). This cessation of plant uptake corresponded with a reduction in the amount of extractable Rb^+ recovered from the soil (Table 3.4).

The decrease in both plant uptake of Rb and the amount of Rb in the soil corresponded with a week of heavy rain (between 8-10-87 and 13-10-87) which initiated drainage flow in the tile drains (Table 3.7) and raised the water table to within 20 cm of the soil surface (D.J. Horne, pers. comm.). Initially it was thought that during these drainage events some of the Rb was leached out of the soil cores. However, when these rainfall events were simulated in the glasshouse study only 0.7% (SEM = 0.68) of the applied Rb was leached out of soil cores by the equivalent amounts of rainfall that fell on the field cores.

The results from the glasshouse trial suggested that leaching losses of Rb due to the October rainfall were likely to have been small. Hence the Rb unaccounted for in the field trial must still have been present in the soil but was in a form that was not extractable with

Table 3.5 The amounts of Rb taken up by the herbage at each harvest date after the injection of 3.3×10^{-3}M RbCl at 0-2.5, 0-14.5 and 12-14.5 cm soil depths.

Soil depth injected	Amount of Rb in herbage (µg Rb core^{-1})							
	Replicate							
Harvest date	1	2	3	4	5	6	mean	CV (%)
0-2.5 cm treatment								
11 Aug 1987	25.5	37.7	1.3	1.6	14.0	25.5	17.60	83
16 Sep	10.4	42.2	0	0	5.7		11.66	151
19 Oct	17.3	50.3	0	0			16.90	140
26 Nov	0	0	0				0	0
4 Jan 1988	0	0					0	
16 Feb	0						0	
11 Mar	0							
% recovery	5	11	0.1	0.1	2	2		
0-14.5 cm treatment								
11 Aug 1987	77.8	88.9	53.7	75.9	45.5	38	63.30	32
16 Sep	186.8	143.0	92.9	62.9	6.8		98.48	71
19 Oct	112.8	56.1	18.9	9.3			49.27	95
26 Nov	0	0	0				0	0
4 Jan 1988	0	0					0	
16 Feb	6.3						6.3	
11 Mar	0						0	
% recovery	5	4	2	2	1	0.4		
12-14.5 cm treatment								
11 Aug 1987	0	0	0	0	0	4.3	0.72	250
16 Sep	0	0	0	0	0		0	0
19 Oct	0	0	0	0			0	0
26 Nov	0	0	0				0	0
4 Jan 1988	0	0					0	
16 Feb	0						0	
11 Mar	0						0	
% recovery	0	0	0	0	0	0.4		

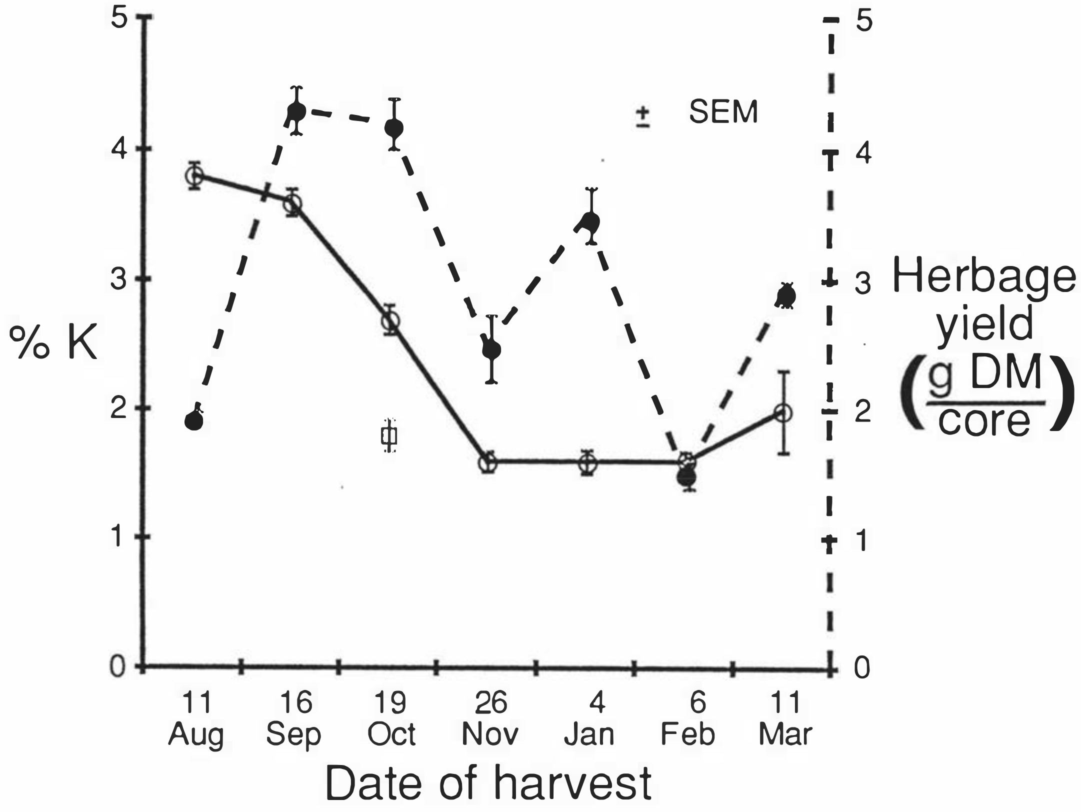


Figure 3.2 Herbage dry matter yield (●) and herbage K^+ concentration (○) at each harvest date. Values are means for all treatments (i.e., 18 replicates in August decreasing to 3 replicates in March). The K^+ concentration of herbage from areas not affected by urine is also presented (□).

Table 3.6 The distribution of pasture root mass in the Tokomaru silt loam.

Depth of soil (cm)	Root weight kg DM ha^{-1}	SEM	%
0-5	4340	331	53
5-10	1850	155	23
10-15	820	86	10
15-20	460	107	6
20-25	410	63	5
25-30	240	40	3
			100 %

Table 3.7 The amount of rainfall, number of rainy days, and amount of predicted drainage during the field trial. Data collected from the DSIR, Palmerston North and D.J. Horne (pers. comm.).

Time period	Rainfall (mm)	No. of rainy days	Predicted drainage (mm)
8 Jul - 11 Aug 1987	46.7	21	30.2
12 Aug - 16 Sep	77.7	14	0
17 Sep - 7 Oct	30.1	9	0
8 Oct - 19 Oct	75	6	3.2
20 Oct - 26 Nov	54.8	16	0
27 Nov - 4 Jan 1988	118.3	20	0
5 Jan - 4 Feb	5.2	3	0
5 Feb - 16 Feb	117.6	11	0
17 Feb - 11 Mar	45.5	16	0

ammonium acetate (e.g., nonexchangeable Rb^+) or available for plant uptake. Whereas it may be possible to extract nonexchangeable Rb^+ from the soil, the extractant would also recover nonexchangeable K^+, which is found in this soil in large quantities (110 mg K g^{-1} soil; Metson, 1980). As described in section 3.2.1, measuring the Rb^+ concentration (3 μg Rb g^{-1} soil) in the extract against this background would be extremely inaccurate

Results from the glasshouse experiment showed that only 65% (SEM = 9.3) of the 5.7 mg Rb applied per core was recovered after a time period of one month. This contrasts with the field experiment where it was possible to extract approximately 100% of the applied Rb after one month from the soil (Table 3.4). Uptake by the herbage over the one month period in the glasshouse experiment did not account for the remaining 2 mg Rb. The reasons for the low recovery of Rb from the glasshouse cores after one month and the field cores after three months remain unclear.

3.3.2 **Conclusions on Rubidium Data**

The results from the first 3 months of the field experiment imply that no significant amounts of Rb were taken up by plants from below the 12 cm depth of soil. However, the usefulness of Rb as an analogue for K in this experiment is uncertain due to the high degree of variation in the data from both the soil and herbage analyses, and the large decrease in the amounts of Rb remaining in the extractable soil pool after the September sampling.

While it was not possible to use the Rb data to achieve the objectives of this experiment outlined in section 3.1.1, the herbage K and exchangeable soil K^+ data from the same experiment provided an alternative data set to study the plant uptake of urine K from different soil depths.

3.3.3 **Fate of Urine Potassium Applied to Soil**

3.3.3.1 *Retention in the soil*

The application of urine to the soil increased the amount of exchangeable K^+ in the soil immediately, particularly in the 0-2.5 and 2.5-12 cm depths (Figure 3.3). This increase in exchangeable K^+ accounted for only 41% of the K applied in the urine, suggesting that a major loss of K occurred during the urine application. The results from the glasshouse experiment showed that immediately after the urine was applied to the soil cores a large amount of K (equivalent to 38-55% of that applied) was collected in the leachate. The volume

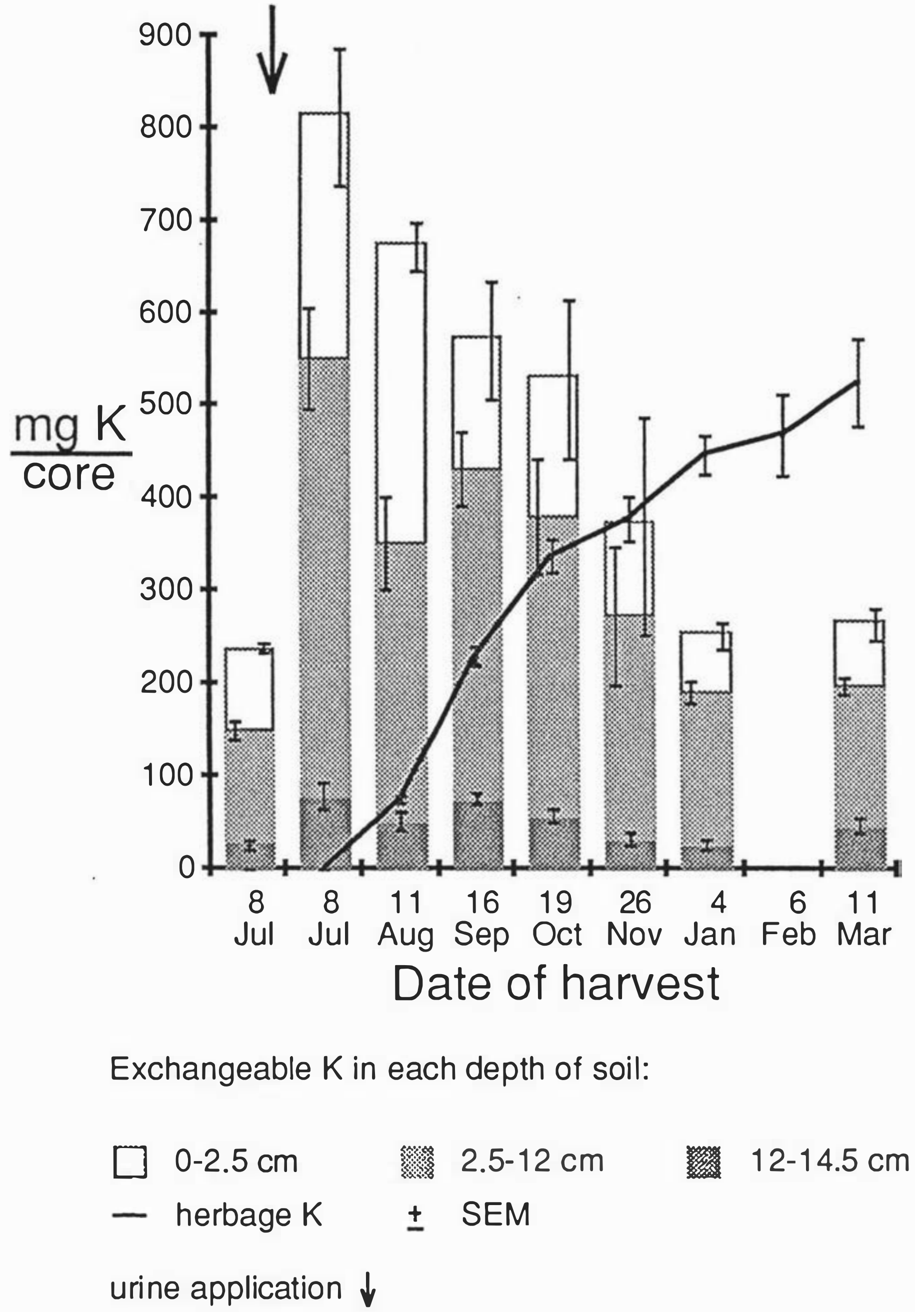


Figure 3.3 Exchangeable soil K^+ and cumulative herbage K at each harvest.

of urine applied (200 ml) represented 0.15 pore volumes, which would have been insufficient for complete piston displacement of the soil water in the core and insufficient for the urine/soil water dispersion front to reach the bottom of the core. The concentration of K^+ in the leachate (2,400-3,500 μg ml^{-1}) more closely resembled the K^+ concentration of the cow urine (6,600 μg ml^{-1}) than that usually found in soil solution (3-30 μg ml^{-1}; section 2.2.4), suggesting that the K in the leachate was derived from the preferential movement of the urine through the soil core via a network of macropores in the soil. The observations from the dye study in the glasshouse experiment confirmed that liquids could move through the soil core via macropores (larger than the 1 mm diameter template pins of the injection apparatus which were still obvious in the soil cores; section 3.2.3) created by plant roots, earthworms and soil fracture planes. Preferential movement of solute through undisturbed cores of Tokomaru silt loam soil has been measured by Kanchanasut et al. (1978).

3.3.3.2 *Uptake by plants*

The fastest rate of K uptake by the pasture plants occurred between the August and October harvests (Figures 3.2, 3.3). This coincided with the fastest pasture growth rates and the period when K^+ concentrations in the herbage were highest (Figure 3.2). The K^+ concentration of herbage from pasture not affected by urine was analysed during October to provide information on the background concentration of K^+ (Figure 3.2). The herbage K^+ concentration from urine treated cores was similar to this background value by the harvest on the 16th of November (Figure 3.2).

The total amount of K that was taken up by the plants in the field experiment was 525 mg (Figure 3.3). This amount of K is similar to both the increase in the amount of soil exchangeable K^+ brought about by the urine application (580 mg K core^{-1}; Figure 3.3), and the difference between the amount of soil exchangeable K^+ at the beginning (after urine application) and end of the experiment (550 mg K core^{-1}; Figure 3.3). Furthermore, results from the glasshouse study showed that only a small amount of K (27.0 mg K core^{-1}, SEM = 4.3) was leached from the simulated rain events, suggesting that leaching losses of urine K retained as exchangeable K^+ in the soil in the field experiment were small. Therefore the amount of K recovered by the herbage during the field experiment was equivalent to the amount of K retained in the soil as exchangeable K^+ after the urine event. Such a result suggests that within a year after urine deposition plant roots will be able to recover the urine K retained in the soil solution and on the soil surface exchange sites from the 0-15 cm depth.

Previous research on the duration of the effect of urine on herbage growth has been based on comparisons of the K content of herbage growing in urine affected and unaffected soil, and changes in exchangeable soil K^+ over time intervals of 6 months or more. From this research it was concluded that urine K affects pasture growth for 6 months on a yellow-brown loam (Saunders and Metson, 1959) and approximately one year on a yellow-grey earth (During and McNaught, 1961). The results from the present study support these conclusions.

Examination of the data in Figure 3.3 in more detail shows that the added urine K increased the amount of exchangeable K^+ mainly in the 0-2.5 and 2.5-12 cm soil depths. Throughout the trial period the quantity of soil K in the top two soil depths decreased by 200 and 320 mg K core^{-1} respectively. The amount of exchangeable K^+ in the 12-14.5 cm depth decreased by only 30 mg K core^{-1}. The pattern of these decreases indicates that most of the plant K was extracted from the upper depths with very little uptake of K from below 12 cm. Such a result confirms the general finding that plants recover the majority of their nutrient requirement from the surface layers of the soil profile, which contain the majority of the plant roots. For example, previous experimental work on the Tokomaru silt loam using ^{32}P and ^{35}S to measure root activity showed that 85 and 55% of the ^{32}P root activity occurred in the 0-4 cm depth of soil in spring and summer respectively, compared with 1 and 10% at the 14-20 cm depth. The root activities for ^{35}S were 74 and 54% from the 0-4 cm depth of soil, and 3 and 12% from the 14-20 cm depth for the respective seasons (Horne, 1985).

Another factor which may have resulted in lower uptake from depths greater than 12 cm in this experiment was that this study measured plant uptake from urine affected soil. The application of urine, which contained both K and N, increased the quantity of soil K down to a soil depth of 12 cm and it was possible that the plant roots in this depth of soil were also well supplied with N. Although K has little influence on root growth, added N can increase root density (Barber, 1984). Therefore localised increases in root density due to the urine N may have enabled the plants to extract their K requirement from the 0-12 cm depth without the need for deeper extraction.

The moisture content of the soil can affect the depth of soil from which nutrients are absorbed by plant roots. The roots of perennial ryegrass swards tend to absorb nutrients from soil depths below 18 cm during periods of time when the top 5 cm of soil dries out (Garwood and Williams, 1967; Newbould et al., 1971). During these periods the roots in the top soil become inactive (Garwood and Williams, 1967). In the field experiment reported in this chapter, the surface layers of soil dried out only in January when climatic conditions were very dry (i.e., only 5.2 mm of rain was recorded over 34 days; Table 3.7). Due to the dry conditions, herbage growth over this period was very small and this, combined with a low K concentration in the herbage (Figure 3.2), resulted in a K uptake that was only 5% of the total K recovered over the whole trial period. Such a result suggests that while plants may absorb water from deep down the soil profile for survival during a dry period, and some K may be absorbed with this water, the amount of K absorbed will be small compared with the total amount of K recovered by the plants during the year.

The decrease in the amounts of exchangeable soil K^+ at each soil depth during the field experiment has been expressed per mg of root weight in each soil depth (Table 3.8). The amount of K taken up by the plants from each soil depth was considered to be the decrease in the amount of soil exchangeable K^+ during the experimental period. The root weights at each depth (0-2.5, 2.5-12 and 12-14.5 cm) were interpolated graphically using the data in Table 3.6. The ratios of mg K taken up by plants : mg of root weight were similar for each soil depth (Table 3.8), which emphasises the dependence of plant K uptake on the amount of roots present (Barber, 1984; section 2.6.1). Of the total weight of roots in the soil profile only a small proportion is found below a depth of 15 cm (Table 3.6). Plant available soil K moves mainly by diffusion over distances of 0.1-15 mm (Barber, 1984). Below the 15 cm depth of soil (where root density is low) the amount of K that comes in contact with, and therefore can be absorbed by, plant roots must be small compared with the upper depths of the soil (where root density is high). It is concluded that the amount of K that is taken up from soil depths below 15 cm must be only a small proportion of the total plant K uptake.

In other field experiments on a free draining sand soil, ryegrass plants recovered ^{42}K used as a tracer from depths of 35 cm (Ozanne et al., 1965). Peterson and Smith (1973) used ^{39}K as a tracer on a silt loam soil and showed that plants could recover K fertiliser placed at depths down to 83 cm. However, they used a deep rooting plant species (lucerne) which may be expected to exploit nutrients at greater soil depths than ryegrass and white clover. Since the plant available K pool at each soil depth was increased by up to 8 fold through the addition of ^{39}K, the results of Peterson and Smith (1973) can not be used to determine the uptake of K from unamended soil. The results of Ozanne et al. (1965) showed significant uptake of ^{42}K from the lower soil depth. It may be, however, that the plant available K pool at the lower depths were small, which could have caused the specific activity of the K taken up from that depth to be high. Data on the size of the soil K pool is not presented by Ozanne et al. (1965) therefore relative root activity for K uptake at each soil depth can not be calculated.

Extrapolation of the results of this experiment to other geographical areas is limited to situations where plant roots are concentrated in the topsoil (i.e., >50% of plant roots in the 0-5 cm depth) and the rainfall distribution ensures that the soil surface dries out for only short

Table 3.8 The ratio of the amount of K absorbed by plant roots (as measured by the change in soil exchangeable K^+ during the experiment) to the weight of roots at each soil depth.

Depth of soil (cm)		Change in soil K (mg K core^{-1})	Weight of roots (mg core^{-1})	Ratio
0-2.5		200	4710	0.042
	SEM	12	359	0.0072
2.5-12		320	6820	0.047
	SEM	29	571	0.0095
12-14.5		30	868	0.035
	SEM	6	91	0.0110

periods of time. Such areas are likely to be the western areas of New Zealand (e.g., Taranaki, Waikato and Manawatu) which have reliable summer rainfall. These areas are the main dairy farming regions of New Zealand.

3.3.3.3 *Influence of rainfall*

Whereas the glasshouse experiment indicated that leaching losses of K due to rainfall in this field experiment were very small, it is unrealistic to extrapolate these results to predict the amount of K that could be leached from urine affected soil when greater amounts of rainfall are applied. While this experiment was carried out over the mid winter/spring/summer period, during other times of the year (e.g., late autumn/early winter) leaching losses of K may be larger. The amount of K leached by rain events from urine treated soil will be examined further in Chapters 5 and 6.

3.4 CONCLUSIONS

Following a urination event in winter on to ryegrass/white clover pasture, 41% of the applied urine K was retained in the top 0-15 cm depth of the Tokomaru silt loam soil as exchangeable K^+. A simulation of the field experiment in the glasshouse indicated that the remainder of the urine K was lost from the top 0-15 cm depth of soil by immediate preferential flow of urine to lower soil depths. Virtually all of the exchangeable K^+ retained in the 0-15 cm depth of soil was taken up by the plant roots over the spring and summer. Thus plant uptake of K is a major mechanism for K conservation in the K cycle.

The majority of the K taken up by plant roots growing in urine affected soil was derived from the 0-12 cm depth of soil in this experiment. The decrease in the amount of exchangeable K^+ at each soil depth throughout the experiment correlated well with the root density at each soil depth. Such results suggest that the recycling of K in a grazed dairy pasture occurs mainly within the top 0-15 cm depth of soil. Further research on the movement of K in a cycle on grazed dairy pasture will need to consider this depth of soil. It is also important to study the factors which influence the retention of urine K in this upper soil depth. For example, K adsorption by urine affected soil and the volume of urine that moves immediately out of the plant root zone by preferential flow are likely to vary with soil properties. These aspects are addressed in Chapters 4, 5, 6 and 7.

In addition, a major loss of K (38-55% of the K applied in the glasshouse experiment) occurred from the 0-15 cm depth of soil after the urination event due to preferential flow. The occurrence and significance of this effect will be researched further in Chapters 5 and 6. The amount of urine that moves preferentially may be affected by soil type, hence in the next chapter the effect of soil type on adsorption of urine K will be examined.

CHAPTER 4

POTASSIUM ADSORPTION BY SOIL AS INFLUENCED BY SOME SOIL CHARACTERISTICS AND THE FORM OF POTASSIUM

4.1 INTRODUCTION

In the previous chapter it was shown that the urine K retained as exchangeable soil K^+, within the effective plant rooting zone of a ryegrass/white clover pasture, was recycled to the above ground herbage within one growing season. It was also shown that on the Tokomaru silt loam the effective rooting depth for K uptake was the 0-15 cm depth of soil (section 3.3.3). Therefore the amount of urine K that can be adsorbed by the soil within 15 cm of the surface is important for conserving and recycling K. Apart from the K^+ concentration in soil solution, the extent to which a soil may adsorb K from solution is determined by the quantity of negative charge on the surfaces of the soil. The amount of negative charge carried by a soil is mostly a function of the amounts and types of clay mineral present in the soil, its organic matter content and pH (section 2.2.3). There is a considerable amount of literature on K adsorption and K leaching in soils (e.g., Munson and Nelson, 1963; section 2.7.1.1) with most studies using potassium chloride (KCl) as a source of added K. Dairy cow urine not only contains K and Cl but also N in the form of urea, which on contact with soil is hydrolysed to ammonium (NH_4^+) and then is subsequently nitrified to nitrate (NO_3^-). Hydrolysis of urea initially increases soil pH and then nitrification decreases soil pH (Doak, 1952; Helyar, 1976). Soil pH has been shown to increase by at least 2 units within 48 hours of urine application (Doak, 1952; Holland and During, 1977). Such an increase in pH could be expected to raise soil cation exchange capacity by 30 me $100g^{-1}$ in soils with large amounts of pH dependent charge (Edmeades, 1982). This increase in negative surface charge may increase K adsorption and subsequently reduce K leaching. In the short term, after urination the high pH in urine spots may be an important influence on the conservation of K returned to the soil in dairy cow urine (c. 20 g of K per urination). Moreover predictions of the amount of K lost from field soils using data obtained with KCl solutions rather than urine, do not include the effects of soil pH changes or changes in NH_4^+ and NO_3^- concentrations and could provide misleading results.

4.1.1 Objectives

To determine whether KCl solutions can be used to simulate dairy cow urine by comparing the extent of K adsorption from two soils contrasting in their surface charge characteristics.

4.2 MATERIALS AND METHODS

4.2.1 Soils

Two topsoils (0-7.5 cm) were used in this experiment; the Tokomaru silt loam and the Egmont brown loam (Table 4.1). These soils have been described in detail by Cowie (1978) and Campbell and Wilde (1970). Both soils are used for intensive dairy farming in New Zealand yet have contrasting chemical properties (Table 4.1). Before use, the soil was air dried and sieved to <2 mm.

4.2.2 Incubation Experiment

Cow urine was collected from the Massey University No. 3 dairy farm during a spring milking. It was frozen immediately and stored until required. A KCl solution was prepared to give the same K^+ concentration as that in the urine. The relevant chemical analyses of this urine and the KCl solution used in this experiment are shown in Table 4.2.

Experimental units each comprised 4 g of soil in an open 10 ml test tube lightly capped with aluminium foil. The soil was moistened to a moisture content of 0.40 g g^{-1} with 1.6 ml of either dairy cow urine or KCl solution. These units were incubated at laboratory temperature (20°C ± 3°C) for up to 106 days. At various times during the incubation, duplicate units of each treatment were removed and extracted with deionised water before extracting exchangeable cations by sequential extraction with 1M sodium chloride (Schofield, 1949). The K removed from soil by these sequential extractions was referred to as water soluble K^+ and adsorbed K^+, respectively. Changes in the adsorption of exchangeable cations were used to indicate changes in soil surface negative charge. Previous work at Massey University (Bolan et al., 1986) using similar soil types incubated with K salts indicated that duplicate samples were sufficient to measure differences between mean values >10%.

The pH of each unit was measured at the end of the water extraction (4 g of soil were shaken with 20 ml of deionised water in 80 ml polypropylene, screw-capped centrifuge tubes for 2 hours on an end-over-end shaker) prior to the tubes being centrifuged (2,000 rpm for 10 minutes using an IEC UV centrifuge) and the supernatant collected after filtration (Whatman No. 5). This extract was frozen until the concentrations of K^+, NH_4^+, NO_3^- and kjeldahl N were determined. The soil residue was shaken with 60 ml of 1M sodium chloride for 30 minutes, centrifuged and the solution filtered. A second extraction with 1M sodium chloride

Table 4.1 Taxonomic details and some characteristics of the Tokomaru silt loam and the Egmont brown loam.

	Tokomaru silt loam	Egmont brown loam
N.Z. soil group	yellow-grey earth	yellow-brown loam
Soil taxonomic classification	Typic Fragiaqualf	Entic Dystrandept/ Typic Hapludand
pH[1]	6.6	5.8
Cation exchange capacity[2] (me $100g^{-1}$)	22	38
Organic carbon[3] (%)	5.2	10.1
Dominant clay minerals vermiculite	illite[4] hydrous feldspar	allophane[5]

[1] in water section 3.2.5.2
[2] section 3.2.5.3
[3] Tabatabai and Bremner (1970)
[4] Pollok (1975)
[5] N.Z. Soil Bureau (1968)

Table 4.2 Chemical analysis of dairy cow urine and KCl solution.

	urine	KCl
K (mg ml^{-1})	1.12	1.12
Cl (mg ml^{-1})	0.27	0.96
N (mg ml^{-1})	1.13	0
pH	8.6	6.1

was carried out and the two extracts combined together and stored in the freezer until the K^+ and NH_4^+ concentrations were measured (section 4.2.4). Finally 60 ml of 0.02M sodium chloride were shaken with the soil for 5 minutes to reduce the concentration of the entrapped solution. After the tubes had been centrifuged, the supernatant was filtered and frozen until the Na^+ concentration was determined (section 4.2.4). The tubes were then weighed to determine the weight of the entrapped solution. The Na^+ adsorbed by the soil was extracted by shaking with 20 ml of 1M ammonium nitrate. This extraction was repeated and the two centrifuged and filtered supernatants were combined and frozen until the Na^+ concentration was measured (section 4.2.4).

4.2.3 Effect of pH on Potassium Adsorption

A separate experiment was carried out to investigate the effect of pH on K adsorption by the soil. Soil pH buffer curves were also derived from this experiment. Ammonium hydroxide was used to increase soil pH and to provide NH_4^+ ions that would be present in high pH urine spots. Sulphuric acid was used to decrease soil pH. Experimental units comprising 4 g of either Tokomaru or Egmont soil were moistened with 1.6 ml of solution containing varying additions (0-500 me kg^{-1} of soil) of either sulphuric acid or ammonium hydroxide. No K was added in these solutions. After 24 hours duplicate units of each treatment were shaken with deionised water (4 g soil in 20 ml of water for 2 hours as above) and the solution pH measured. Then the samples were centrifuged, filtered and the filtrate retained for water soluble K^+ analysis (section 4.2.4). The residue soil was shaken twice with 60 ml of 1M sodium chloride for 30 minutes to extract adsorbed K^+. The two extracts were separated from the soil as above, combined and the concentration of K^+ in the combined extract was measured (section 4.2.4).

4.2.4 Chemical Analysis

The K^+ and Na^+ concentrations in the extracts were determined separately by atomic emission in the presence of either 0.2% caesium chloride (for K^+) or 0.2% potassium chloride (for Na^+). Ammonium and NO_3^- concentrations in solution were measured by the continuous flow, bubble segmented autoanalyser methods of Technicon (1976) and Downes (1978). Kjeldahl N was measured after 1 ml of solution was digested with 5 ml of kjeldahl mixture (2.5 dm^3 of sulphuric acid + 250 g of potassium sulphate + 2.5 g of selenium powder) at 350°C for 3 hours. After cooling the digest was diluted and the NH_4^+ concentration measured by a continuous flow, bubble segmented autoanalyser (Technicon, 1973).

4.3 RESULTS AND DISCUSSION

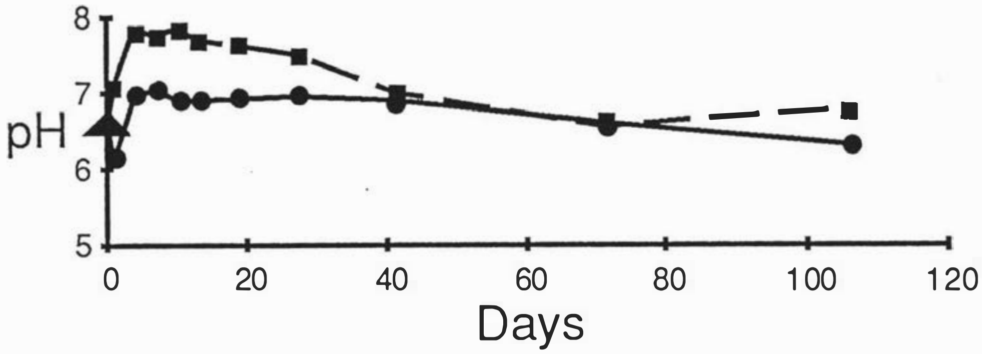
4.3.1 pH and Cation Adsorption

Within 24 hours after incubation with urine there was a rapid rise in soil pH (Day 1; Figure 4.1) in both the Tokomaru and Egmont soils as the urea hydrolysed to increase the concentration of adsorbed and water soluble NH_4^+ in the soil (Figures 4.2, 4.3). The magnitude of this pH increase was only half that reported by Doak (1952) and Holland and During (1977), probably due to the lower N content of the cow urine used in the present experiment.

Upon the addition of KCl the pH of both soils decreased (Day 1; Figure 4.1) compared to the initial soil pH measured in water. This may simply be due to an increased ionic strength of the soil solution caused by KCl addition (Black and Campbell, 1982). Then over the next six days of the experiment, an increase in soil pH was measured in the KCl treatment of both soils (Figure 4.1). For the KCl treated Tokomaru soil this increase in pH was accompanied by an increase in exchangeable NH_4^+ of 86 μg N g^{-1} (Figure 4.2) and water soluble NH_4^+ of 100 μg N g^{-1} (Figure 4.3). If this NH_4^+ (186 μg N g^{-1}) came from the mineralisation of organic matter stimulated by rewetting the soil, then for a net increase of one mole of NH_4^+, one mole of OH^- would also be produced (Helyar, 1976). Using the soil pH buffer curve (Figure 4.4) it was calculated that the amount of OH^- ions produced in this manner accounted for approximately half of the pH increase. [The rest of the pH change cannot be accounted for, however, it is common that pH buffer curves generated from equilibrating soils with acid and alkali over short periods of time (1 day) do not simulate soil pH changes which occur over longer periods of time (6 days). For instance, it has been observed that protons and hydroxyl ions added to generate pH buffer curves are sometimes consumed in reactions other than charge balancing and pH changes (Parker et al., 1979)]. A similar increase in exchangeable NH_4^+ could have accounted for the pH increase of the KCl treatment in the Egmont soil (Figure 4.1). However, due to the large background variation in concentrations of adsorbed and water soluble NH_4^+ between replicates of the Egmont soil, it was difficult to detect a net increase in NH_4^+ over the period of the pH increase (Figures 4.2, 4.3).

In the Egmont soil there was a decrease in pH between days 7 and 13 (Figure 4.1). Such a pH change might have been expected due to nitrification of NH_4^+ to NO_3^-, which produces 2 moles of H^+ per mole of N converted (Helyar, 1976), but no increase in extractable soil NO_3^- could be measured during this period (Figure 4.5).

(a) Tokomaru



(b) Egmont

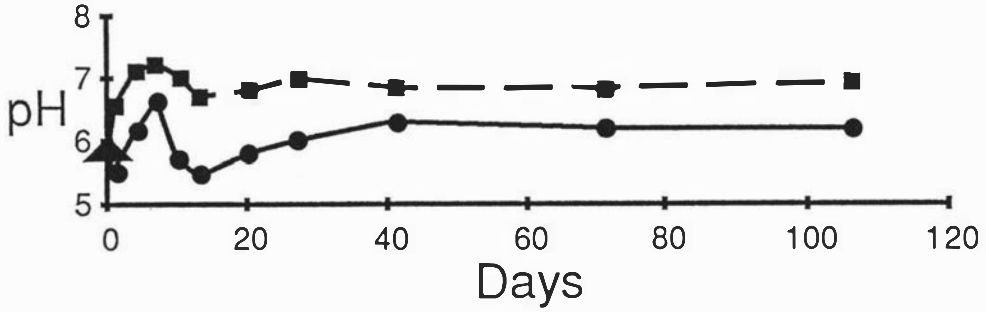
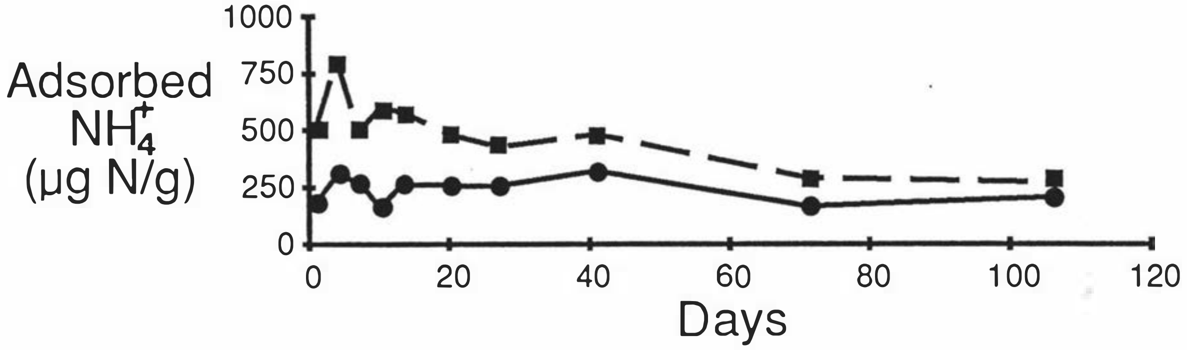
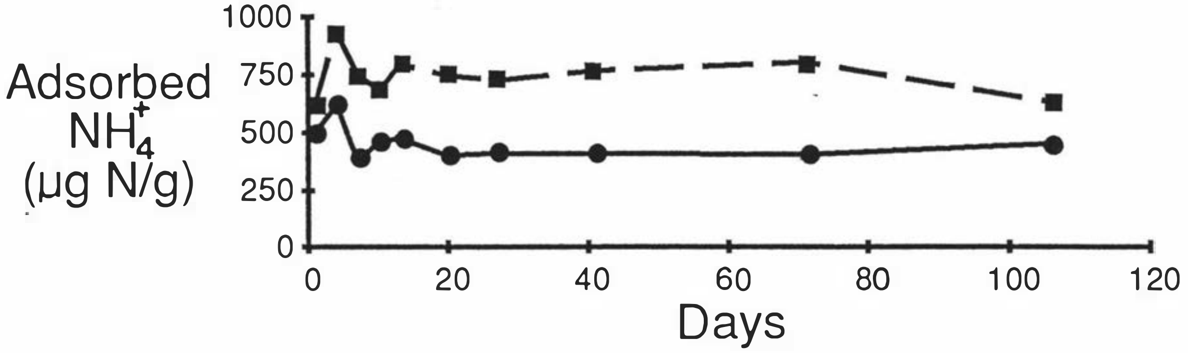


Figure 4.1 Soil pH before (▲) and after incubation with urine (■) or KCl (●). Values are means of duplicate experimental units (mean deviation <2% except for final point of Tokomaru KCl treatment data which showed a 3% mean deviation).



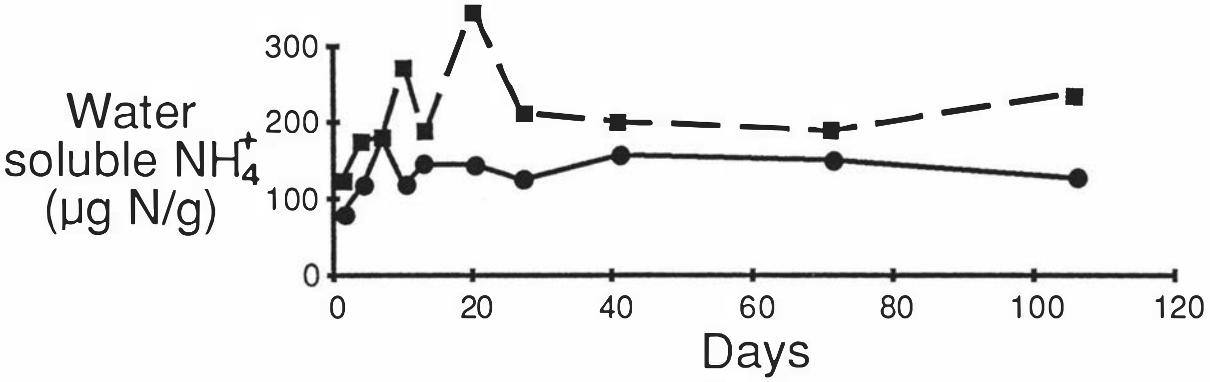
(a) Tokomaru



(b) Egmont

Figure 4.2 The amount of 1M NaCl extractable NH_4^+ (adsorbed) in soil incubated with urine (■) or KCl (●). Values are means of duplicate experimental units (mean deviation Tokomaru soil <15%, and Egmont <25%).

(a) Tokomaru



(b) Egmont

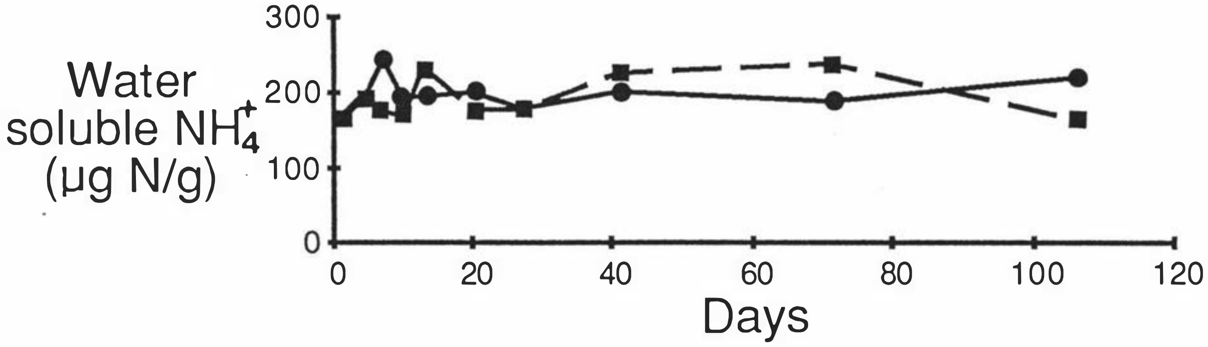


Figure 4.3 The amount of water soluble NH_4^+ extracted from soil incubated with urine (■) or KCl (●). Values are means of duplicate experimental units (mean deviation <25%).

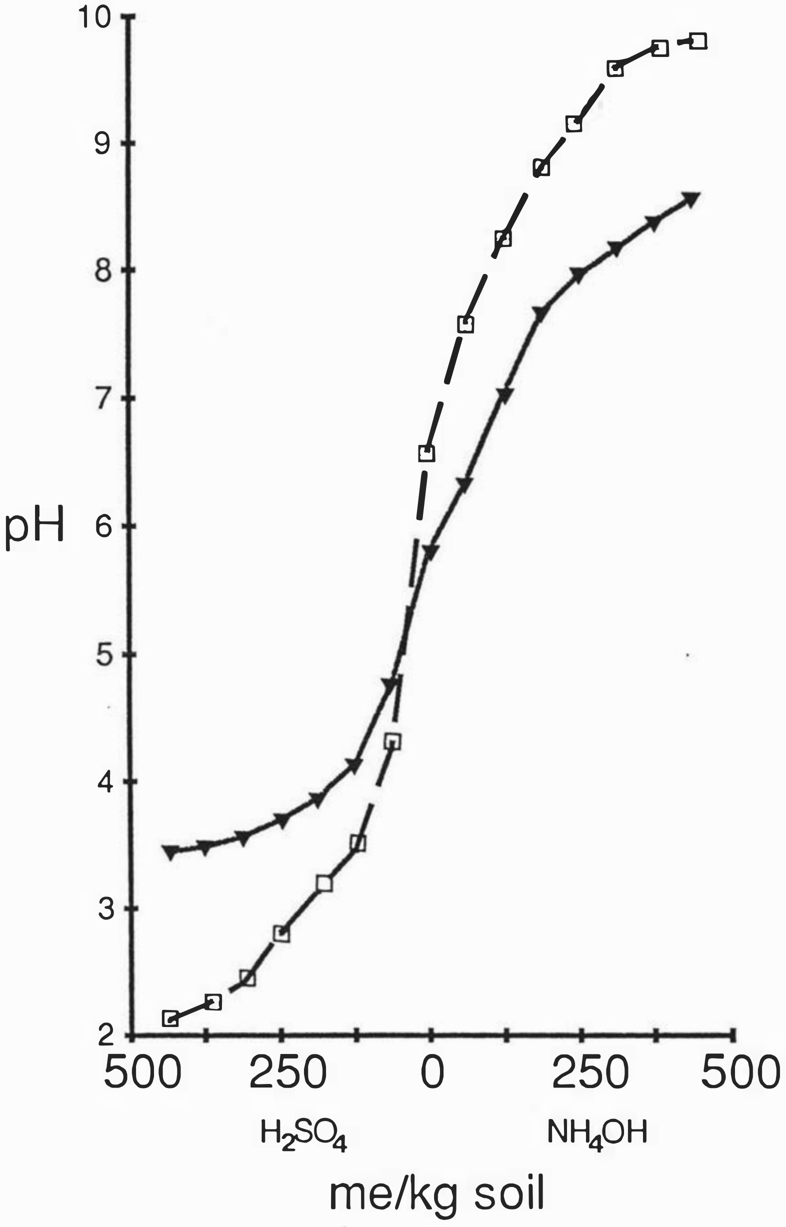
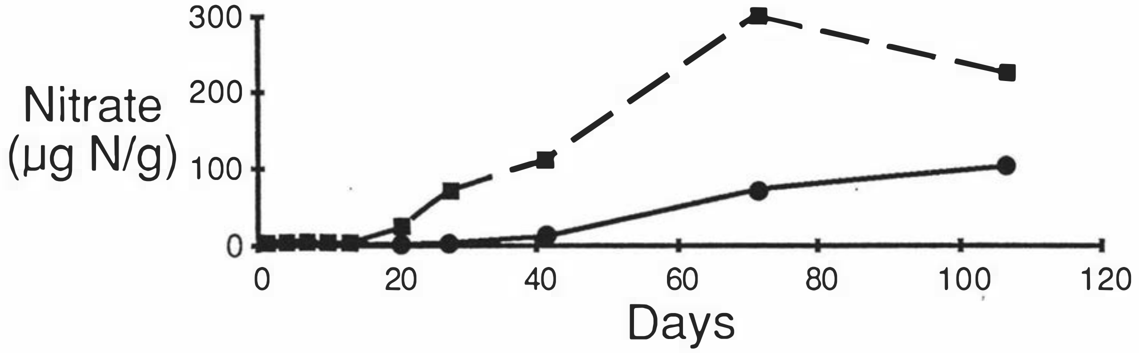


Figure 4.4 Soil pH buffer curves for Tokomaru (□) and Egmont (▼) soils. Values are means of duplicate experimental units (mean deviation <5%).

(a) Tokomaru



(b) Egmont

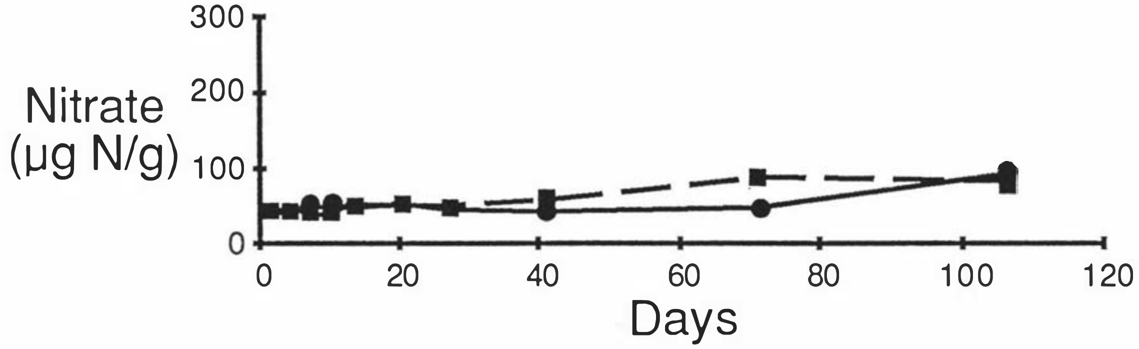


Figure 4.5 The amount of water soluble NO_3^- extracted from soil incubated with either urine (■) or KCl (●). Values are means of duplicate experimental units (mean deviation <10%).

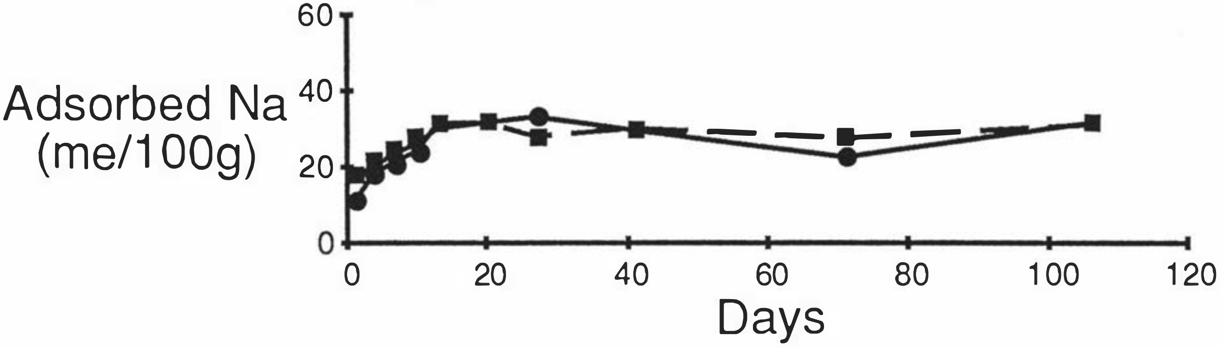
Although the reason for these fluctuations in soil pH (Figure 4.1) could not all be accounted for, the pH increases were associated with expected increases in surface negative charge (as measured by Na^+ adsorption; Figure 4.6) for all soil treatments except the Egmont KCl treatment. The changes in Na^+ adsorption for the KCl treated Egmont soil over the first 7 days of the experiment can not be explained but were consistent between replicates.

In the Tokomaru soil the difference in pH between the urine and KCl treatments decreased with time until after 41 days both treatments had a similar pH (Figure 4.1). In contrast, for the Egmont soil, the pH of the urine treatment was consistently higher throughout the duration of the experiment (i.e., up to 106 days). The duration of the effect of the urine treatment on increasing soil pH will be discussed further in section 4.3.2.

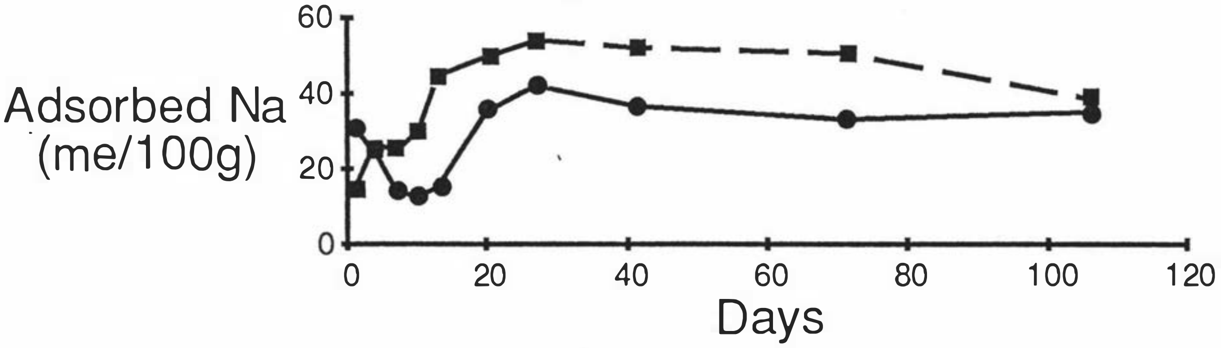
The higher pH produced by the urine treatment compared to the KCl treatment of the Egmont soil resulted in a higher amount of Na^+ adsorption (Figure 4.6). This increase in cation adsorption is due to the OH^- ions produced during urea hydrolysis causing deprotonation of the pH dependent charge sites which result in an increase in negative surface charge as explained by Gast (1977). Such sites are associated with the allophane and organic components which are present in greater amounts in the Egmont soil (Table 4.1). Although there was a difference in soil pH between the urine and KCl treatments over the first 41 days on the Tokomaru soil (Figure 4.1) this did not result in a difference in Na^+ adsorption (Figure 4.6) presumably due to the small amount of pH dependent negative charge in the Tokomaru soil (Bolan et al., 1986). These changes in cation adsorption (Figure 4.6) caused by increases in soil pH are of a similar magnitude to those reported by Edmeades (1982) on other yellow-brown loam and yellow-grey earth soils.

The increase in surface negative charge (as measured by Na^+ adsorption) at the higher soil pH in the Egmont soil also resulted in an increase in the adsorption of K^+ and therefore a lower amount of water soluble K^+ (Figure 4.7). Similar to the Na^+ adsorption results (Figure 4.6) there was little difference in the amount of water soluble K^+ between the two treatments on the Tokomaru soil.

Since the Tokomaru and Egmont soils have different native amounts of K, expressing the data as a ratio of water soluble : adsorbed K^+ allows for comparison between these soils of the effect of pH on the relative distribution of K^+ between the solution and soil surface. A decrease in this ratio, such as occurred with increasing pH on the Egmont soil (Figure 4.8), represents an increase in the amount of adsorbed K^+. Applying urine to the Egmont soil lead to higher soil pHs (Figure 4.1) and lower water soluble : adsorbed K^+ ratios (Figure 4.8) than

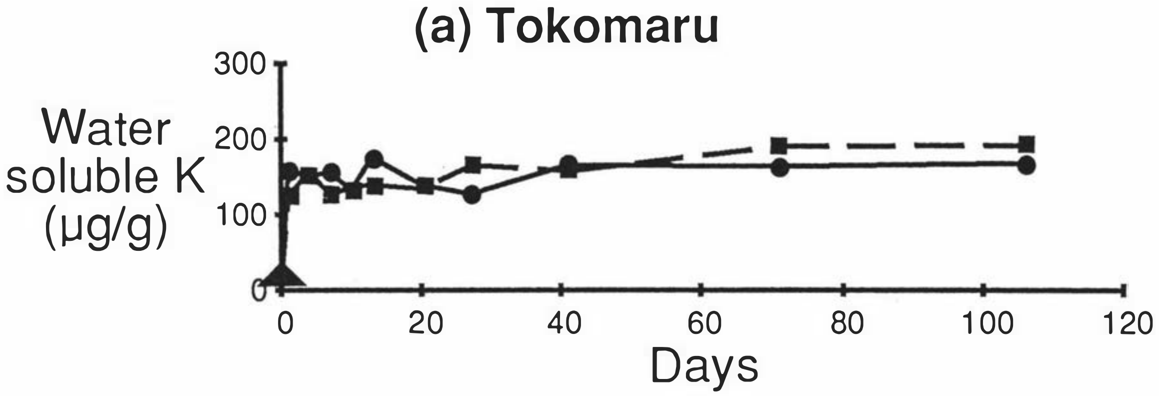


(a) Tokomaru



(b) Egmont

Figure 4.6 The amount of Na^+ adsorbed from a solution of 1M NaCl by soil previously incubated with urine (■) or KCl (●). Values are means of duplicate experimental units (mean deviation <10%).



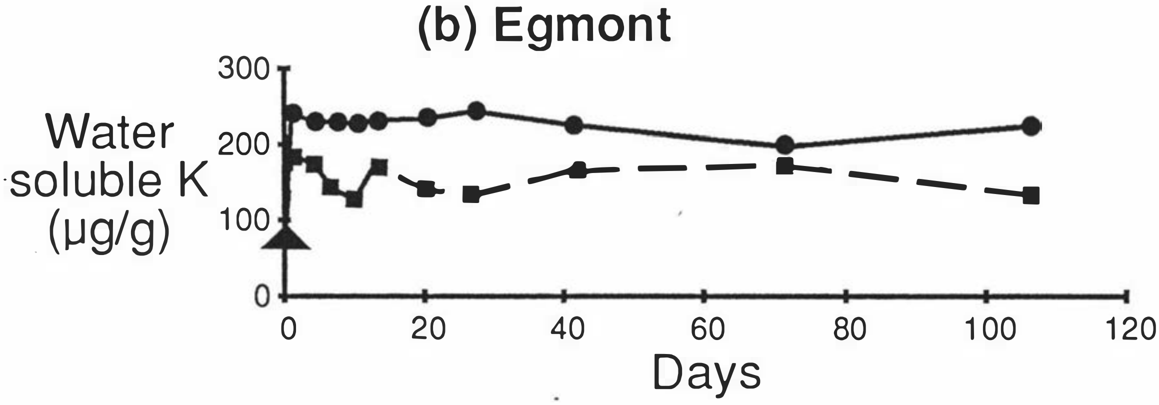
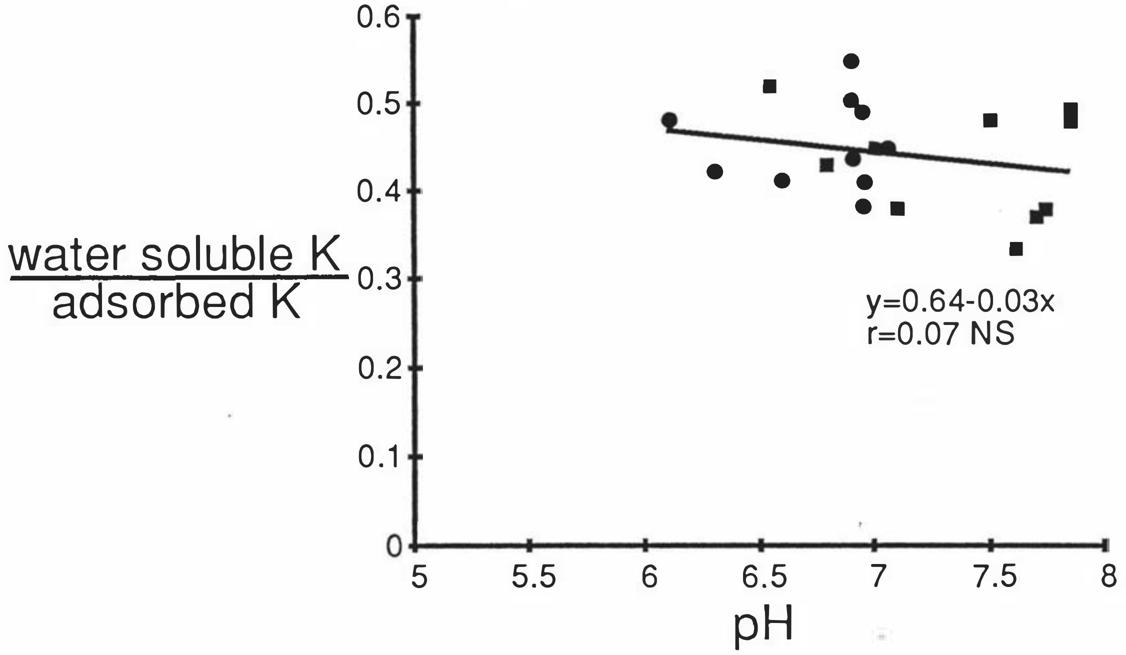


Figure 4.7 The amount of water soluble K^+ extracted from soil before (▲) and after incubation with urine (■) or KCl (●). Values are means of duplicate experimental units (mean deviation <10%).

(a) Tokomaru



(b) Egmont

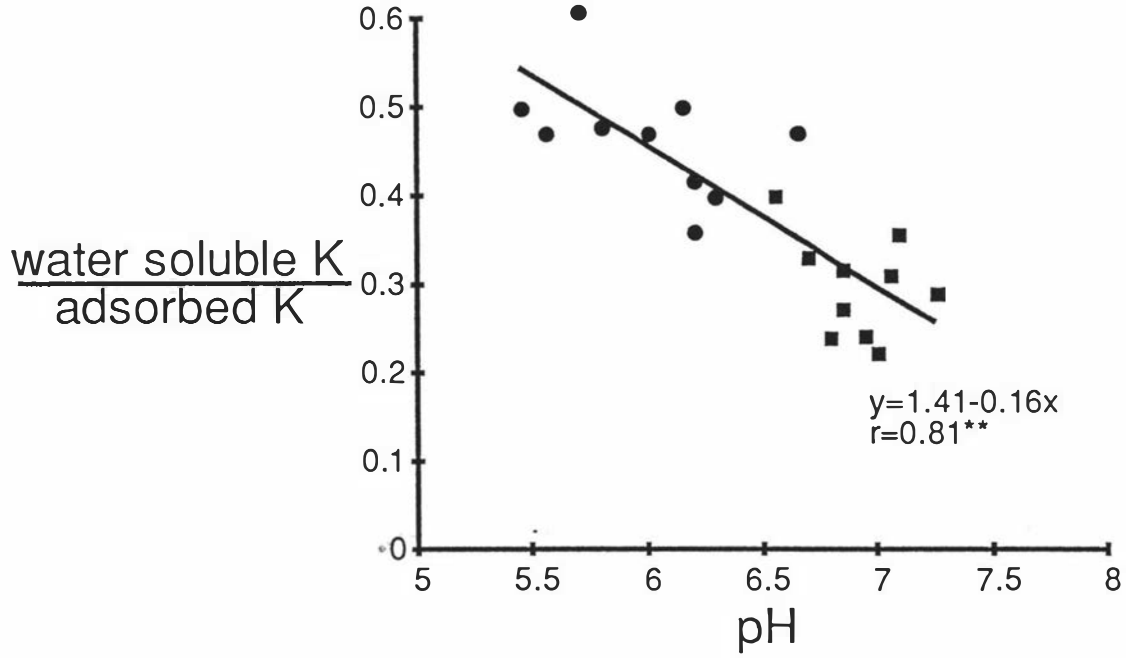


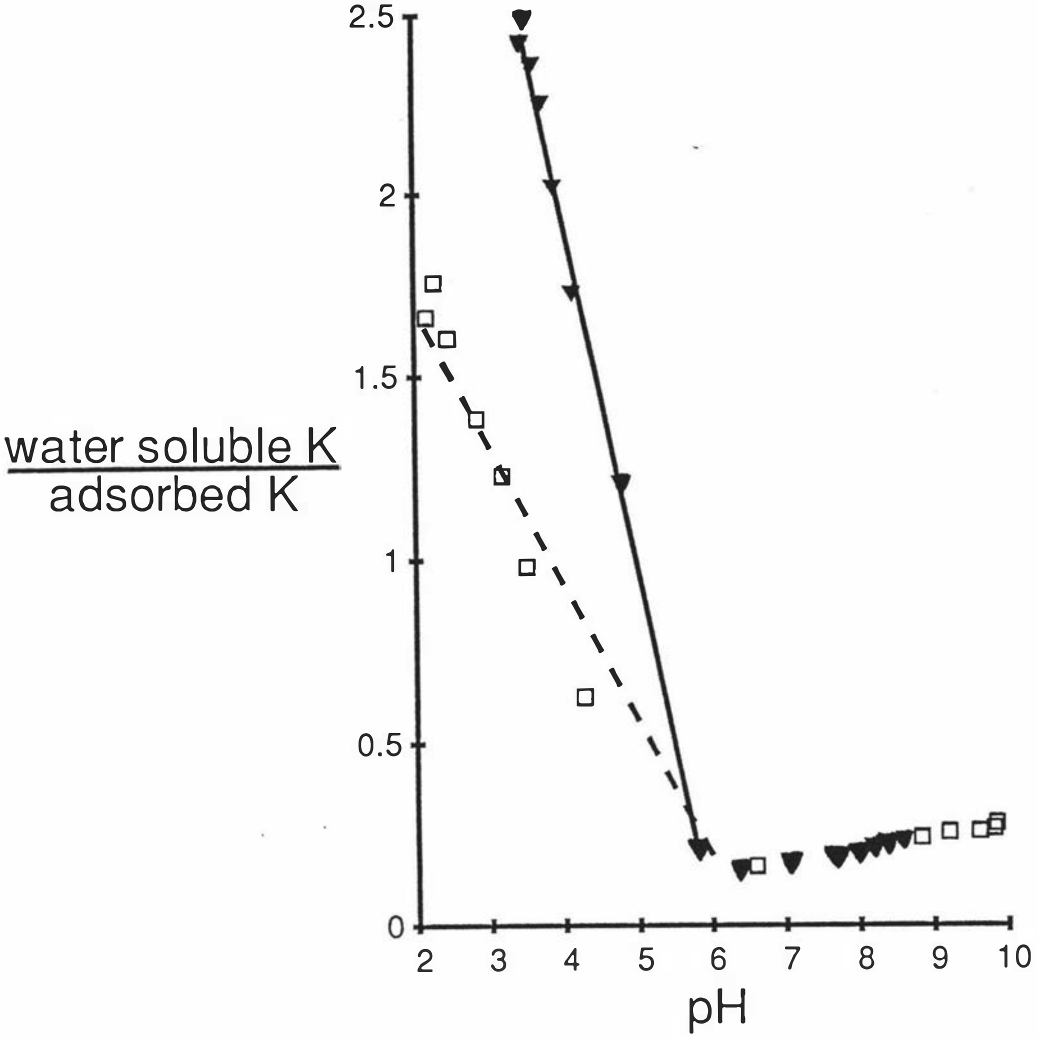
Figure 4.8 The relationship between pH and the ratio of water soluble K^+ : adsorbed K^+ in soils treated with urine (■) and KCl (●). Values are means of duplicate experimental units (mean deviation <10%).

when KCl was applied. The same treatments on the Tokomaru soil did not result in large differences in the size of the ratio of water soluble : adsorbed K^+ (Figure 4.8). The increase in adsorbed K^+ with increased pH on the Egmont soil has also been measured on another yellow-brown loam soil (Patua soil) by Bolan et al. (1988).

The effect of a change in pH on the ratio of water soluble : adsorbed K^+ is clearly shown in the second incubation experiment. Figure 4.9 shows the effect of incubating these two soils with varying amounts of sulphuric acid or ammonium hydroxide on the distribution of native soil K between water soluble and adsorbed forms. As the pH increased from 2 to 6 there was a significant increase in the proportion of K^+ that was adsorbed on the soil and a corresponding decrease in water soluble K^+. The changes in the ratio of water soluble : adsorbed K^+ between pH 2 and 6 were linearly related to soil pH on both soil types (Figure 4.9), but the gradient was steeper for the Egmont soil presumably reflecting a greater increase in cation adsorption sites as pH increased.

Above pH 6 the ratios of water soluble : adsorbed K^+ changed little on both soils when compared to the change that occurred below pH 6 (Figure 4.9). This presumably reflects a relatively constant minimum concentration for water soluble K^+ in these soils partly maintained by the displacement of adsorbed K^+ by NH_4^+ added as ammonium hydroxide. When soil solution K^+ concentrations were increased 3-10 fold by urine or KCl addition (in the Egmont and Tokomaru soils respectively; Figure 4.7), and where the soil pH values were greater than 6, then the ratio continued to decrease rapidly on the Egmont soil (Figure 4.8) but not on the Tokomaru soil.

The urine treatment not only added K but also N (Table 4.2) to the soil in the form of urea. After the urea had hydrolysed to NH_4^+ there would have been competition between the NH_4^+ and K^+ for the adsorption sites in the soil treated with urine. In the Tokomaru soil, the presence of NH_4^+ in the urine treatment appeared to have negligible effect on K^+ adsorption resulting in similar concentrations of water soluble K^+ (Figure 4.7) and similar proportions of adsorbed K^+ (Figure 4.6) in both the KCl and urine treatments. For the Egmont soil, all of the NH_4^+ from the urine appears to have been adsorbed by the soil surface as there was little difference in the amount of water soluble NH_4^+ between the urine and KCl treatments (Figure 4.3). Despite the presence of this adsorbed NH_4^+ there was still increased K^+ adsorption with increased pH in the Egmont soil (Figure 4.8). This indicates that in the Egmont soil, which contained more pH dependent charge than the Tokomaru soil, the increased soil pH after the urine application created enough CEC to retain both the urine K added and the NH_4^+ formed from the urine.



Fitted lines for the pH range 2-6

□ Tokomaru
$y = 2.41 - 0.37x$ $r = 0.962^{**}$

▼ Egmont
$y = 5.79 - 0.96x$ $r = 0.998^{**}$

Figure 4.9 Effect of changing soil pH, through incubation with either sulphuric acid or ammonium hydroxide, on the distribution of natural soil K between water soluble K^+ and adsorbed K^+. Values are means of duplicate experimental units (mean deviation <10%).

Although the ratio of native water soluble : native adsorbed K^+ was similar in both soils when the soil pH was >6 (Figure 4.9), at field pH values the Egmont soil had higher concentrations of native K^+ in soil solution (Day 0; Figure 4.7). More native K can be expected to be leached from the Egmont soil because leaching losses are directly related to the amount of K in soil solution. When considering native K only, similar proportions of the total soil K could be leached from each soil. However, applying urine to each soil temporarily increased both soil pHs, but surface negative charge was increased significantly compared with the KCl treatment only in the Egmont soil which decreased the solution : adsorbed K^+ ratio (Figure 4.8). In these circumstances, especially when soil pH remains high, a smaller proportion of the added urine K can be expected to be leached from the Egmont soil than the Tokomaru.

4.3.2 Duration of pH Effect on Cation Adsorption

The increase in cation adsorption on the Egmont soil treated with urine could be expected to last for as long as the increased pH was sustained. In the incubation experiment reported here, an increased pH was evident for the 106 days of the experiment (Figure 4.1). This effect is attributed to a lack of nitrification in the Egmont soil over this period as very little NO_3^- was measured in any of the water extracts (Figure 4.5). In fact, throughout the experiment most of the urine-N on the Egmont soil appeared to be present as adsorbed NH_4^+, the amounts of which remained constant throughout the experiment at about 750 $\mu g\ g^{-1}$ soil (Figure 4.2). In contrast, on the Tokomaru soil nitrification appears to have started between 13 and 20 days after the incubations began, as there was a significant increase in NO_3^- (25 $\mu g\ g^{-1}$) appearing in the water extract on day 20 (Figure 4.5), with a corresponding decrease in soil pH (Figure 4.1) and adsorbed NH_4^+ amounts (Figure 4.2). This effect is consistent with the work of Doak (1952) who found an increase in nitrification after 12 days in a soil incubation study involving sheep urine.

The lack of nitrification in the Egmont soil is inconsistent with other incubation experiments which have shown that very high rates occur on yellow-brown loam soils (Sarathchandra, 1978; Steele et al., 1980). Since nitrification proceeded as expected in the Tokomaru soil, and the Egmont soil was subjected to the same experimental conditions, this result is difficult to interpret. The incubation was carried out in an open test tube and was unlikely to have generated anaerobic conditions which would have inhibited nitrification.

Although the pH effect on cation adsorption was apparent on the Egmont soil for the 106 days of this experiment, it is difficult to extrapolate this finding to the field situation where the controlled environmental conditions of the laboratory are absent. Evidence from published field trials on New Zealand soils including a yellow-brown loam shows that the increase in pH resulting from urine application is only evident for 30 days in the topsoil (Doak, 1952; Holland and During, 1977; Field et al., 1985).

4.3.3 Implications for Potassium Leaching

Only a few studies have compared K leaching from KCl and urine. These have all involved pot experiments or lysimeter studies using New Zealand soils with considerable pH dependent charge (yellow-brown loam and yellow-brown pumice soils). Research on the yellow-brown loam soils has shown twice as much K leached from KCl compared with urine (Saunders and Metson, 1959; Davies and Hogg, 1960). Saunders and Metson (1959) attributed their result to better pasture growth on the urine treatment, while Davies and Hogg (1960) predicted a preferential loss of NH_4^+ over K in the urine treatment although no NH_4^+ analyses were carried out. In a later paper Davies et al. (1962) again referred to the experiment of Davies and Hogg (1960) and suggested that the lower leaching loss of K from urine may have been due to the presence of more bicarbonate and less chloride in the urine. Hogg (1981) found little difference in the proportion of K leached from a yellow-brown pumice soil treated with either dairy cow urine or KCl in a study using 1 m deep lysimeters. However, the amounts of K leached were small (<4% of that applied) making precise comparisons between the treatments difficult.

In the experiments reported in this study with Egmont soil, the KCl treatment always resulted in a greater proportion of the added K remaining in solution than when the soil was treated with urine (Figure 4.7). This suggests that a higher proportion of the K from KCl would be lost by leaching compared with the K from dairy cow urine on soils with large amounts of pH dependent charge.

Furthermore, results from this incubation experiment, suggest that the different effects of urine and KCl are partly related to changes in soil pH and its subsequent effect on surface charge due to urine addition which would result in greater adsorption of both K and NH_4^+.

4.4 CONCLUSIONS

Similar proportions of K from dairy cow urine and KCl were adsorbed by the Tokomaru soil. In contrast, in the Egmont soil more K was adsorbed from the dairy cow urine than from the KCl. This difference between the two soils was caused by a greater increase in negative surface charge in the Egmont soil compared to the Tokomaru soil, due to the increase in soil pH following the urine application. In the field situation this implies that similar proportions of K could be leached from either K source on the Tokomaru soil, but on the Egmont soil reduced leaching losses could be expected from urine.

The results of this experiment show that data on K leaching from KCl applied to soil should not be used to predict losses of K from a grazed pasture, as in the grazed dairy pasture a large proportion of the K being recycled is in the form of urine. Furthermore, there may be differences between soil types of contrasting mineralogy in the proportion of urine K that is adsorbed by the negative exchange sites. In the following chapter (Chapter 5) the fate of dairy cow urine applied to four soils of contrasting mineralogy will be studied.

CHAPTER 5

FATE OF DAIRY COW URINE APPLIED TO INTACT SOIL CORES

5.1 INTRODUCTION

In Chapter 3, the results from a field experiment demonstrated that a ryegrass/white clover sward on the Tokomaru silt loam did not recover significant amounts of urine K from soil depths below 15 cm. Chemical analysis of the soil immediately following urine deposition indicated that approximately 41% of the applied urine K could be accounted for as exchangeable soil K^+ in the top 15 cm of the soil profile. The results from an accompanying glasshouse study (section 3.2.3) with intact soil cores showed that immediately after application urine flowed through soil macropores to beyond the 15 cm soil depth. The amount of K in this leachate was similar (38-55% of the applied K) to that unaccounted for in the field cores (59% of applied K). If this immediate leaching of urine K is widespread then for a farm on the Tokomaru silt loam grazing 3 cows ha^{-1} (which would return 220 kg K ha^{-1} yr^{-1} on to the paddock as urine) 130 kg K ha^{-1} yr^{-1} could be lost by this route.

The fate of urine K after deposition on soil has been investigated previously in New Zealand by several authors using a wide range of experimental techniques (Table 2.6). As discussed in section 2.6.4.1, results from these experiments on the fate of urine K were extremely variable due to the variety of experimental techniques, soil types and urine K concentrations that were involved. Only the experiments of Saunders and Metson (1959) and Hogg (1968; 1981) can be considered to provide information that is directly relevant to the field situation. These researchers found that the fate of urine K was as follows: 1-7% was lost by leaching, 51-90% was taken up by the plants and 0-33% was retained by the soil (Table 2.6). These experiments were confined to yellow-brown loam and yellow-brown pumice soils. As shown in Chapter 4, such results should not be extrapolated to other soil types with contrasting mineralogy to the yellow-brown loam and yellow-brown pumice soils because differences in K^+ adsorption can occur between soil types (section 4.3.1). In addition the physical movement of urine through soil may be different for soils of contrasting texture, structure and bulk density. There is a need, therefore, to study the leaching losses of dairy cow urine K from a range of naturally structured, New Zealand soils that vary in their chemical and physical properties.

If, as indicated by the results from Chapter 3, significant amounts of urine K are lost immediately by flow through the soil macropores, the concentration of K^+ in the urine will influence the amount of K lost to the lower soil depths. Measurements of the K^+ concentration in dairy cow urine have shown a tremendous range from 0.35 to 1.56% (Table 2.2). The range is caused by a number of factors including differences between individual cows, and seasonal and daily fluctuations due to diet and temperature (Larvor and Brochart, 1959; Hutton et al., 1967; St Omer and Roberts, 1967; Paquay et al., 1969; Betteridge et al., 1986). Little information is currently available to indicate what effect different K^+ concentrations in urine have on the amounts of K leached or adsorbed by the soil and the proportions of the adsorbed K that are recycled via plant uptake.

This chapter (Chapter 5) reports on a closely controlled experiment designed to study the fate of urine K of different concentrations immediately following a urine deposition on intact soil cores. Four soils were used which contrasted in their chemical and physical properties. The duration of the experiment was equivalent to the interval between grazing events (30 days) on a typical dairy farm in the Manawatu/Taranaki region during the milking season.

The experiment was carried out in a controlled-climate chamber using late August/September climatic conditions for the Manawatu/Taranaki region. These conditions were chosen to represent a time when the leaching of K in the field was expected to be high, because of high soil moisture content and relatively high rainfall. Furthermore the amount of urine K deposited on to the soil can be expected to be high because the K content in the herbage is highest in the early spring (section 2.6.1.2) and pasture growth rates are rapid (section 2.3.1). Hence during late August/September if pasture utilisation by the grazing dairy cows is high, then up to 1.5 kg of urine K ha^{-1} day^{-1} can be deposited on to the soil.

5.1.1 Objectives

The objectives of this study were to:

i. determine the amounts of K that are leached, retained by the soil and taken up by plants over a 30 day period following an application of dairy cow urine to intact soil cores.

ii. compare the various fates of urine K between four soils of different chemical and physical properties.

iii. provide information on whether the fate of urine K is affected by the K^+ concentration in urine.

5.2 MATERIALS AND METHODS

5.2.1 Establishment of Experiment

5.2.1.1 *Sites of soil collection*

The four soils used in this study were a yellow-grey earth, a recent soil, a yellow-brown loam and a yellow-brown sand. The yellow-grey earth contains the micaceous clay minerals illite and vermiculite which are capable of being both a source and a sink for K, respectively (2.2.1). The recent soil has illite as its dominant clay mineral. The yellow-brown loam lacks the K retaining, 2:1 layer, clay minerals, but contains allophane which is thought to be capable of adsorbing K (Sticher, 1972; Metson, 1980). The yellow-brown loam has a large amount of pH dependent negative charge which is expected to result in a greater adsorption of K from dairy cow urine (section 4.3.1). The yellow-brown sand has very low amounts of clay minerals present. Further details of the chemical and physical properties of these soils are given in Table 5.1.

For each soil type, a soil core collection site was selected on established dairy pasture. Cores were taken from areas of high quality pasture. At three sites ryegrass and white clover were the dominant species. At the Pukepuke site the sward contained paspalum as well as ryegrass and white clover. Care was taken to avoid areas of the sward which had obviously been recently affected by urine or dung. The details of these sites and their history are given in Table 5.2.

Twenty-two intact cores were collected from each site. The cores were collected by driving metal cylinders (15 cm diameter x 15 cm depth) into the soil and then digging these out carefully. Each cylinder was pushed in to a depth of 14 cm; a 1 cm high rim was allowed to avoid the loss of liquids that were later applied to the surface. Fifteen cm was chosen as the appropriate depth for the cores because the experiment in Chapter 3 showed that ryegrass/white clover pastures on the Tokomaru silt loam recovered insignificant amounts of K from below a depth of 15 cm. The diameter of the cores was 15 cm, which was considered to be adequate to provide a representative sample of the patterns of soil macropores in the soils used (Bouma, 1980).

Table 5.1 Some physical and chemical characteristics of the 0-7.5 cm depth of the soils used. The methods used to measure soil pH, CEC and exchangeable K^+ are described in sections 3.2.5.2, 3.2.5.3, 5.2.6.2.

	Tokomaru	Manawatu	Egmont	Pukepuke
N.Z.soil group	yellow-grey earth	recent	yellow-brown loam	yellow-brown sand
Classification	Typic Fragiaqualf	Dystric Eutrohrept Inceptisol	Entic Dystrandept/ Typic Hapludand	Mollic Psamnaquent
Soil type	Tokomaru silt loam	Manawatu fine sandy loam	Egmont brown loam	Pukepuke black sandy loam
Dominant clay minerals	illite[1] vermiculite	illite[2]	allophane[3] hydrous mica	amorphous[4] material
Bulk density ($Mg\ m^{-3}$)	1.1	1.2	0.8	1.3
pH	6.4	5.7	5.6	6.1
CEC (me %)	22	16	38	22
Exchangeable K^+ (me %)[5]	0.23	0.55	0.25	0.15
Organic carbon[6] (%)	5.2	3.4	10.1	6.0
Olsen P[7] ($\mu g\ g^{-1}$)	24	48	19	33

[1] Pollok (1975)
[2] N.Z. Soil Bureau (pers. comm.)
[3] N.Z. Soil Bureau (1968)
[4] Steele (1972)
[5] 0-15 cm depth of soil
[6] Tabatabai and Bremner (1970)
[7] Olsen et al. (1954)

Table 5.2 The locations and the background information on the sites from which the soils were collected.

	Tokomaru	Manawatu	Egmont	Pukepuke
Location from Palmerston North	5 km East	2 km East	145 km North/ West	25 km West
Map reference	T24 312866	T24 321891	N129 825375	S24 071867
Annual rainfall[1]	1000	1000	1260	910
Pasture dry matter production (kg DM ha^{-1})[2]	16,000	18,000	18,000	15,000
Potential carrying capacity (su ha^{-1})[3]	26	30	30	25
Annual fertiliser history over last three years[4]	200 kg ha^{-1} 15% S-super[5]	nil	350 kg ha^{-1} 30% K-super[6]	375 kg ha^{-1} 30% K-super

[1] N.Z. Meteorological Service (1973)
[2] Water & Soil Division, Ministry of Works & Development; N.Z. Land Resource Inventory
[3] From kg DM ha^{-1} assuming 1 su = 550 kg DM and 90% utilisation
[4] Information supplied by farmers
[5] S-super = sulphurised single superphosphate
[6] K-super = potassic single superphosphate

Additional soil samples were collected from around each core in order to determine the initial amounts of water soluble and exchangeable cations by chemical analysis (section 5.2.6.2). Soil pH (section 3.2.5.2) and cation exchange capacity (section 3.2.5.3) were also measured on a bulked sample of these cores. On the basis of these analyses 18 of the 22 cores were selected to provide a set of cores with similar initial exchangeable K^+ status.

5.2.1.2 *Preparation of intact cores*

Each core was prepared for the experiment by covering the bottom with nylon gauze secured with a rubber band. The volume of soil in each core was estimated as the difference between total cylinder volume and the volume of soya beans that were required to fill the core to the top of the cylinder. The volume/weight relationship of soya beans was calibrated using a measuring cylinder. The cores were placed in trays of distilled water for 10 days, after which time they were drained for 24 hours and weighed. This weight was defined as the "core capacity" weight.

Each core was then placed on a 14 cm diameter plastic container inside a 20 cm diameter plastic container as shown in Figure 5.1. This enabled leachate to be collected from the inner 154 cm^2 of the soil core and the outer 23 cm^2. This precaution was taken in case the urine and rain applications flowed down the interface between the soil and the cylinder. To prevent evaporation of the leachates a plastic bag was placed around the outside of the plastic containers and secured around the cylinder with a rubber band.

Neither the weight of leachate collected per unit cross-sectional area of soil core nor the chemical composition of the leachates collected separately from the inner and outer portions of the soil cores indicated that the applied liquids were flowing preferentially down the cylinder walls. This is consistent with the findings of M.S. Smith et al. (1985). Therefore, in the following discussion of the results (section 5.3) the data from the inner and outer leachates have been combined together.

Throughout the experiment the cores were kept in a controlled-climate chamber which provided a 10 hour day at 14°C and a 14 hour night at 6°C. The relative humidity was 80%. These climatic conditions are typical of the Manawatu/Taranaki region in late August/September (New Zealand Meteorological Service, 1973).

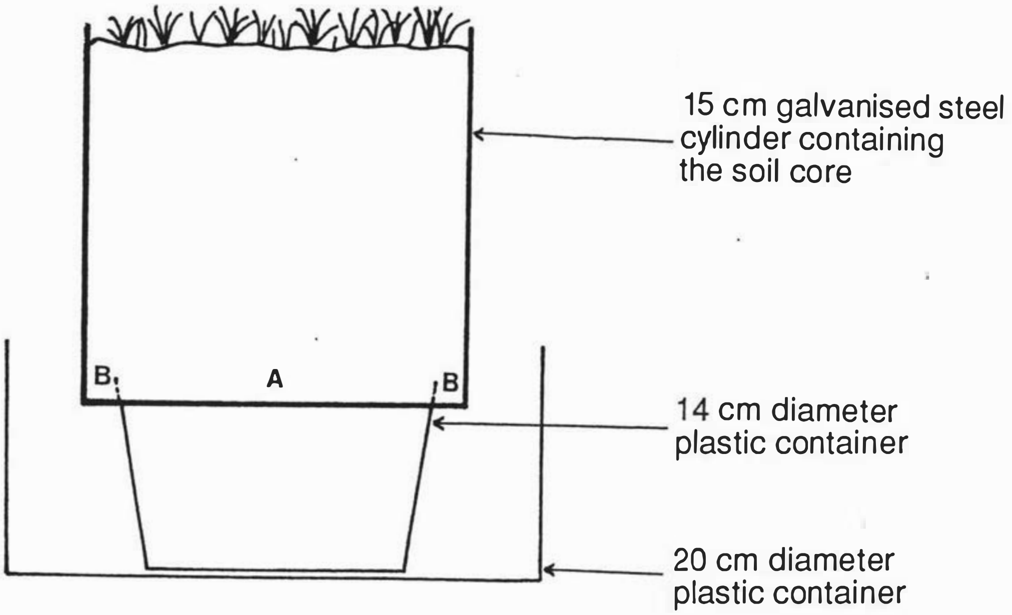


Figure 5.1 A diagrammatic representation (not to scale) of the system used to collect leachate from the inner area A (154 cm^2 in the 14 cm diameter plastic container) and the outer area B (23 cm^2 in the 20 cm diameter plastic container) of the soil cores.

5.2.1.3 *Experimental treatments*

Three concentrations of dairy cow urine were applied: 0% K (deionised water), 0.6% K and 1.3% K. These treatments are subsequently referred to as water, low urine and high urine treatments. The urine was collected from the Massey University No. 3 dairy farm at a morning milking in February 1986. Immediately after collection, the urine was frozen until required for the experiment. Sixteen hours prior to the application to the soil cores the urine was removed from the freezer and allowed to thaw at room temperature. Some of the urine was diluted with deionised water to create the low urine treatment. The results of the chemical analysis of the thawed urine used in these treatments is given in Table 5.3.

5.2.2 **Experiment Design**

The 4 soils and 3 urine treatments were organised in to a 4 x 3 factorial design with 6 replicates. Within each soil type the cores were ranked according to soil K status, from highest to lowest, and then allocated randomly to each replicate. The ranking involved use of the soil exchangeable K^+ contents that had been measured in the soil samples taken from around each core in the field (section 5.2.1.1).

5.2.2.1 *Urine application*

The herbage on the cores was trimmed to 2 cm above the soil surface and the urine was applied at the rate of 200 ml per core. This represents a field application rate of 11 dm^3 m^{-2}, which was similar to that used by Hogg (1968; 1981). On the third day after application the leachate was removed, weighed and a sample retained for chemical analysis. The leachate sample was filtered (Whatman No. 5 filter paper) and 2 drops of toluene were added before storing the sample in the freezer prior to the measurement of the K^+, Ca^{++}, Mg^{++}, NO_3^--N, NH_4^+-N and Cl^- concentrations (section 5.2.6.1).

5.2.2.2 *Simulated rainfall*

The mean annual rainfall for a 30 day period during late August/ September in the Manawatu/Taranaki region is 90 mm (New Zealand Meteorological Service, 1973). This rainfall was simulated as nine "rain" days, each of 10 mm, which began three days after the urine application. On each "rain" day, two rain events of 5 mm were applied. The interval between the rain events was 4 hours. Each rain event comprised 90 ml of deionised water applied from a wash bottle. This procedure was repeated every third day until 90 mm of simulated rain had been applied.

Table 5.3 The chemical composition of the urine samples (for methods of analysis see section 5.2.6.1).

	Low urine	High urine
Total salts	27 mg ml^{-1}	58 mg ml^{-1}
Potassium	6 mg ml^{-1}	13 mg ml^{-1}
Calcium	13 $\mu g\ ml^{-1}$	31 $\mu g\ ml^{-1}$
Magnesium	75 $\mu g\ ml^{-1}$	188 $\mu g\ ml^{-1}$
Sodium	11 $\mu g\ ml^{-1}$	26 $\mu g\ ml^{-1}$
Kjeldahl Nitrogen	6 mg ml^{-1}	12 mg ml^{-1}
Nitrate	<0.01 $\mu g\ ml^{-1}$	<0.01 $\mu g\ ml^{-1}$
Phosphate	<0.01 $\mu g\ ml^{-1}$	<0.01 $\mu g\ ml^{-1}$
Sulphate[1]	250 $\mu g\ ml^{-1}$	500 $\mu g\ ml^{-1}$
Chloride	3 mg ml^{-1}	5 mg ml^{-1}
pH	8.5	8.5

[1] Johnson & Nishita (1952)

5.2.3 Herbage Analysis

Three days after the final rain event, when the leachates were collected for the last time, the herbage was harvested at ground level. It was dissected in to grass and clover fractions, dried at 65°C for 24 hours and weighed. The dried herbage was ground finely and then digested as described in section 3.2.4. The concentration of K^+ and Na^+ in these samples was determined by atomic emission in the presence of 0.2% caesium chloride (K^+) and 0.2% potassium chloride (Na^+). The Ca^{++} and Mg^{++} concentrations were determined by atomic absorption in the presence of 0.2% strontium nitrate.

5.2.4 Root Analysis

At the end of the experiment two cores from each treatment were selected for examination of the roots. These were divided into two soil depths (0-7.5 and 7.5-15 cm) and two separate samples of roots were picked out from each depth. These samples were dried at 65°C for 24 hours, shaken to remove any adhering soil, then ground. The K^+ concentration in the roots was measured using the same method as for the herbage (section 3.2.4). Approximately 20 g of soil from each depth was retained for chemical analysis. The remainder of the root material was washed to remove the soil then dried at 65°C for 24 hours and weighed.

5.2.5 Soil Analysis

The cores were oven dried for three days at 65°C after which time it was possible to remove the cylinder from around the soil core. Each core was sliced into two depths which were left to dry at room temperature. After sieving (<2 mm), the soil pH (section 3.2.5.2) and the concentrations of water soluble and exchangeable K^+, Ca^{++}, Mg^{++} and Na^+ in the soil samples (section 5.2.6.2) were determined.

5.2.6 Chemical Analysis

5.2.6.1 *Urine and leachate analysis*

The samples of urine and leachate were diluted with deionised water as required to give K^+ and Ca^{++} concentrations in the range of 1-10 ppm, and Mg^{++} concentrations of 0.1-1 ppm. The concentrations of these cations in solution were measured as described in section 5.2.3.

The pH of the leachates was measured directly without any sample preparation using a combination glass electrode and a Radiometer PHM61 pH meter.

The concentrations of NO_3^--N, NH_4^+-N and Cl^- in the leachates were measured by the continuous flow, bubble segmented, autoanalyser methods described by Downes (1978), Technicon (1976) and Allen (1974) respectively. The kjeldahl N and total phosphate (PO_4^{3-}-P) concentrations in the urine were determined after digesting 1 ml of sample with kjeldahl digest mixture as described in section 4.2.4. After digestion the extracts were diluted and the NH_4^+-N and PO_4^{3-}-P concentrations were measured by the autoanalyser methods of Technicon (1973) and Twine and Williams (1971).

5.2.6.2 *Water soluble and exchangeable cations in soil*

Four g of dried sieved soil were shaken on an end-over-end shaker with 20 ml of deionised water in 50 ml polypropylene, screw-capped centrifuge tubes for 2 hours. The tubes were centrifuged (10,000 rpm for 5 minutes using a Sorvall RC5C centrifuge) and the supernatant was collected after filtering (Whatman No. 5 filter paper). The K^+, Na^+, Ca^{++} and Mg^{++} concentrations in these extracts were determined as described earlier (section 5.2.3).

The concentrations of exchangeable cations remaining in the soil after the water extraction were determined using the method of Thomas (1982). Enough water was added to the soil sample in the centrifuge tube to make the total water content up to 10 ml, and 10 ml of 2M ammonium acetate (pH 7) were added. The tubes were shaken end-over-end for 30 minutes prior to centrifugation and collection of the supernatant (A) after filtering as above. A second extraction followed. This involved shaking the residual soil with 20 ml of 1M ammonium acetate (pH=7). Again, the supernatant was recovered after centrifugation and filtration and was added to the previous extract (A). The volume of the combined extracts was adjusted to 100 ml and the concentration of K^+, Na^+, Ca^{++} and Mg^{++} measured as before (section 5.2.3).

5.2.7 **Statistical Analysis**

To establish whether there were significant differences between treatment means an analysis of variance was conducted. The least significant differences between means were calculated using the technique described by Steel and Torrie (1960, p. 106). Treatment effects which were significant have been denoted as follows: $P<0.05$ = *, $P<0.01$ = **, $P<0.001$ = ***. Non significant effects have been denoted by NS. The results of the root analysis, which were obtained from duplicate observations only, were not treated in this manner.

5.2.8 Incubation Experiment

A soil incubation experiment was carried out to determine the extent of recovery of applied K from the four soil types used in the intact core experiment. Topsoil was collected from the 0-7.5 cm depth of the soil profile at each of the four sites described in section 5.2.1.1 and was air dried and sieved (<2 mm). Experimental units comprising 4 g of soil were moistened to a moisture content of 0.40 g g^{-1} with 1.6 ml of either water, dairy cow urine or KCl solution. The K content of the urine and KCl solutions were both 1.2 mg K ml^{-1}. These units were incubated at laboratory temperature ($20^{o}C \pm 3^{o}C$) for up to 30 days. At 0, 10, 20 and 30 days after the incubation began, triplicate units of each treatment were extracted for water soluble and exchangeable K^{+} using the methods described in section 5.2.6.2.

5.3 RESULTS AND DISCUSSION

5.3.1 The Fate of Urine Potassium In and the Overall Recovery of Potassium From the Soil Cores

The amounts of K that were lost from the soil cores during the treatment application, leached during the subsequent rainfall events, taken up by the herbage and which remained in the roots and soil at the end of the experiment are shown in Table 5.4. Figure 5.2 shows this data depicted as a percentage of the amount of K applied. Over all soil types and urine concentrations 29-50% of the applied K remained in the soil at the end of the 30 day experimental period. Immediately following treatment application a major loss (5-39%) of K occurred due to leaching. In contrast, the percentages of K leached after the simulated rainfall (2-6%) and taken up by the herbage (1-9%) were relatively small. The majority of K taken up by the plants (63-87% of total K in roots and herbage) remained in root tissue. However, the size of this K pool may have been overestimated due to soil adhering to the roots. The soils of this experiment contained 2-3 times more K per unit of weight than the roots.

Whereas 95-108% of the applied K was recovered from the low urine treatment, significantly less ($P<0.01$) was recovered from the high urine treatment (71-79%). It was suspected that the K not accounted for in the high urine treatment had been retained in a nonexchangeable form. However, the clay mineralogy of the soils suggested that this was an unlikely explanation for all the soil types, particularly the Pukepuke soil (Table 5.1). To determine whether there had been an increase in the amount of K in the nonexchangeable form soil samples were extracted with boiling nitric acid following the method of Metson (1968).

Table 5.4 The amounts of leachate, soil and plant K measured following the application of urine of different concentrations (water, low and high K concentration) to intact cores of different soils (mg K core^{-1}).

Urine treatment Soil type	Leached from urine application	Leached during 90mm of simulated rain	Herbage uptake	Roots	Gain in soil K	Total K recovered
Water						
Tokomaru	0.5	1	77	134	-45	
Manawatu	2	8	75	227	-13	
Egmont	3	7	41	108	11	
Pukepuke	0.4	2	61	101	-44	
Low urine						
Tokomaru	325	25	103	223	516	1192
Manawatu	476	51	85	154	392	1158
Egmont	186	46	112	479	579	1402
Pukepuke	370	52	66	324	374	1186
High urine						
Tokomaru	645	77	53	91	1023	1889
Manawatu	861	106	84	154	718	1923
Egmont	135	79	59	253	1246	1772
Pukepuke	382	150	33	222	1197	1984
SED	142.8	13.2	15.7	$	126.8	211.6
Significance of main effect:						
soil	*	***	***		*	NS
urine	***	***	***		***	***

$ Mean deviation of two replicates <40%

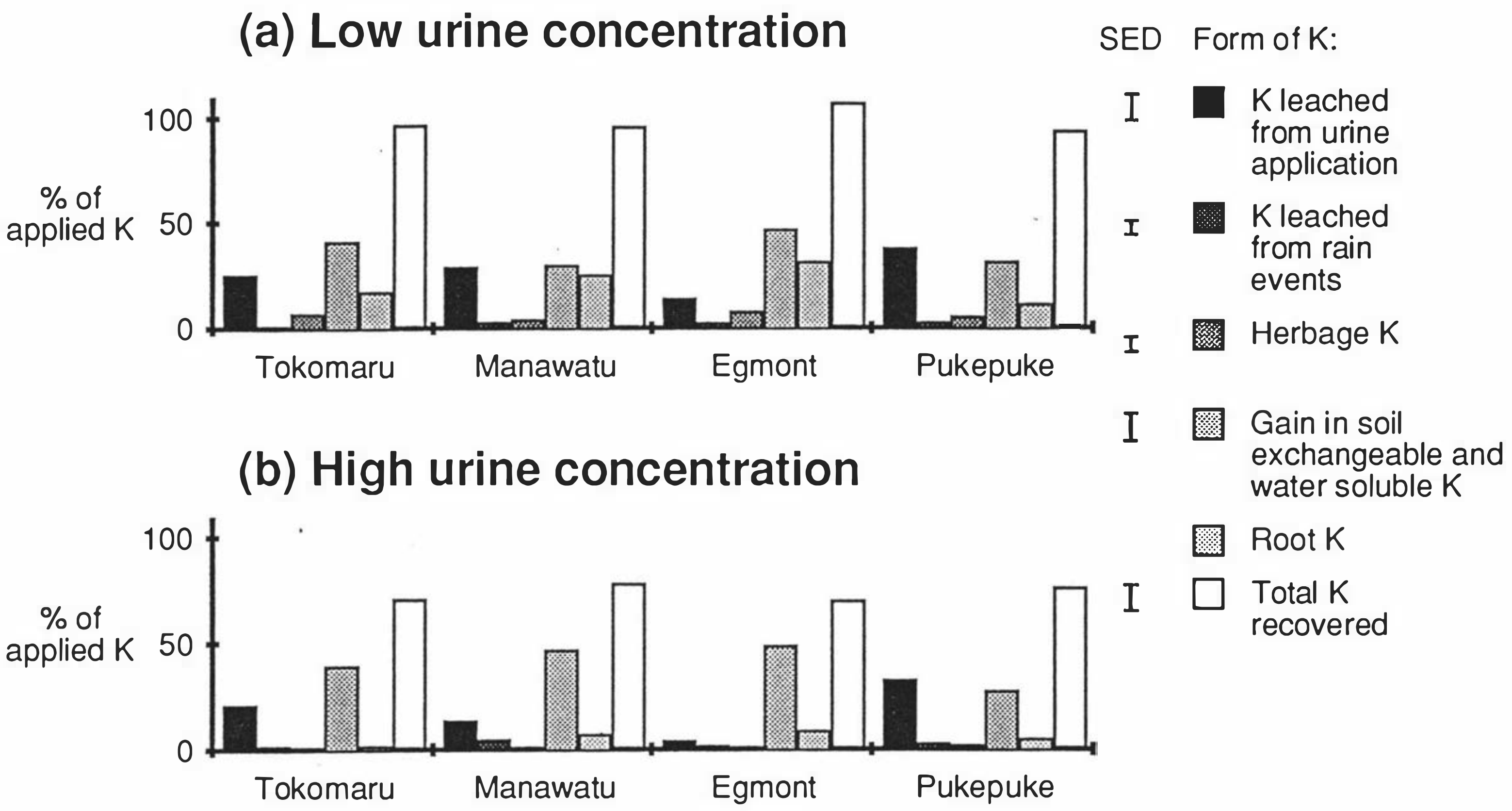


Figure 5.2 The amounts of leachate, soil and plant K present following the application of urine of different concentrations to intact cores of different soil types. The amounts of K are expressed using the formula [(K in urine treated soil - K in water treated soil) x 100 / K applied in urine]. Values are means of six replicates except for the data for the amount of K in roots which are based on two replicates.

This, however, extracted amounts of K similar to those extracted by the ammonium acetate method used previously. There are other methods of extracting nonexchangeable K^+ but these can also remove some of the structural K^+ (Metson, 1980; McLean and Watson, 1985), large amounts of which are found in these soils (Metson, 1980). The quantity of urine K unaccounted for was less than 5% of the total soil K. With such a large amount of native soil K it would be difficult to measure precisely any gain in nonexchangeable K^+ arising from the urine application.

The soil incubation experiment described in section 5.2.8 was used to determine whether or not urine-K or KCl-K was retained in nonexchangeable soil forms. There were no significant decreases in the recovery of exchangeable K^+ over the 30 day period of the incubation, therefore only the results for the samples extracted on day 0 and day 30 are presented (Figure 5.3). At day 0 and at day 30 it was not possible to recover 100% of the applied K on all soil types (Figure 5.3). Up to 20% of the applied K could not be recovered from the soils by extraction with ammonium acetate. Therefore, it is likely that the urine K unaccounted for in this experiment was present in the soil in a form which was not recoverable by the ammonium acetate extraction. Hogg (1968) and Saunders and Metson (1959) also failed to recover 100% of the applied K in their experiments. Hogg (1968) found that 5-17% of the K applied as KCl was not recoverable in an exchangeable form after incubation for one month with a yellow-brown loam soil. Saunders and Metson (1959) used intact soil cores to determine the fate of urine K applied to a yellow-brown loam soil. They recovered only 68-73% of the applied K after measuring the total amount of K leached, taken up by the plants and adsorbed by the soil. They tried to account for the remaining 27-32% of the applied K by extracting the soil with boiling nitric acid, but were unable to extract any more K from the urine treated soil than from the control soil. In addition, they checked for loss of plant sap during herbage drying, and K adsorption on the equipment used; no discernible K loss was apparent. However, Saunders and Metson (1959) felt that the discrepancy between applied and recoverable K did not invalidate their results.

5.3.2 Immediate Loss of Potassium From the Treatment Applications

During the first three days following urine application to the soil cores, between 135 and 861 mg K (5 and 39% of the K applied as urine) were collected in the leachate (Table 5.4; Figure 5.2). When water only was applied to the cores losses of K were very low in comparison (Table 5.4), indicating that virtually all the K leached during these three days originated from the urine. The amount of urine applied (200 ml) represented between 0.11 and 0.15 of a pore volume depending on the soil type. This volume would have been insufficient

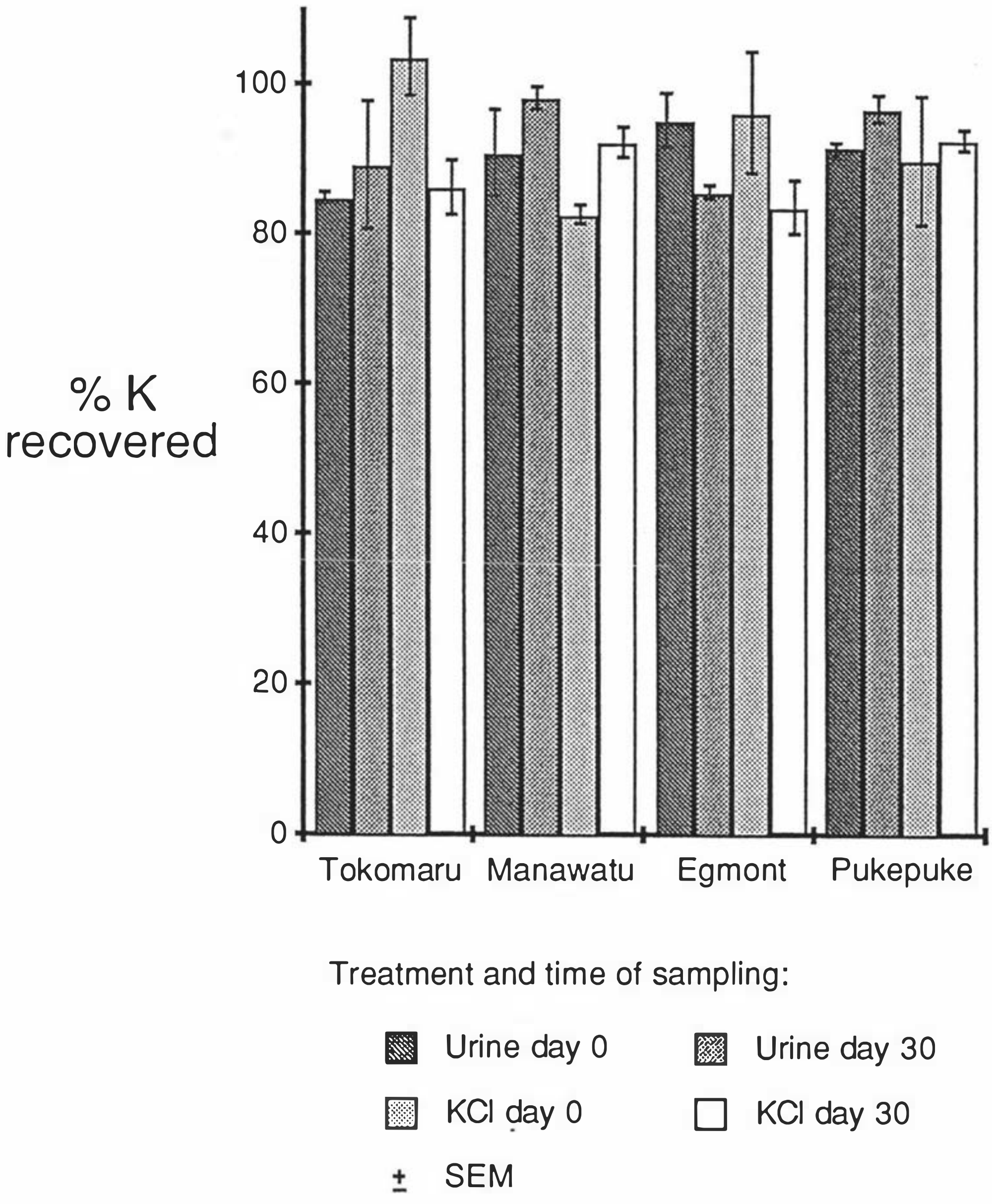


Figure 5.3 The percentages of applied K recovered as the sum of water soluble plus exchangeable soil K^+ immediately after (day 0) and at 30 days after the incubation of four different soils with urine or KCl. Values are means of two replicates.

for complete piston displacement of the native soil water. Therefore the K in the leachate must have originated from preferential flow of urine through a network of macropores in the soil core. Preferential flow of urine through intact soil cores has already been shown to occur on the Tokomaru silt loam in Chapter 3 (section 3.3.3.1), and has been recorded on other soil types by Elrick and French (1966), McMahon and Thomas (1974), Bouma et al. (1981), Tyler and Thomas (1981), White et al. (1984), M.S. Smith et al. (1985) and Francis (1986).

5.3.2.1 *The influence of soil chemical and physical properties*

There were differences between soils in the proportion of urine K appearing in the leachates resulting from preferential flow of urine with losses of 5 and 15% from the Egmont soil, 15 and 30% from the Pukepuke, 23 and 27% from the Tokomaru and 34 and 39% from the Manawatu, for the high and low urine treatments, respectively (Figure 5.2). There were also differences in the leachate weights that were collected between the soils (Appendix I). However, there does not appear to be any relationship between the mean weight of leachate collected and the mean amount of K leached for each treatment (Figure 5.4). This indicates that soil type appears to have had an effect on urine K retention.

Part of the explanation for the different K retentions between the soils may be the way that the urine passed physically through the soil cores. If all the urine had moved through the core by rapid preferential flow then the majority of the K would have bypassed negatively charged soil surfaces which are capable of K adsorption. However, if the preferential flow of the urine through the soil had been slower and more tortuous then a greater amount of the urine K could have been retained by the soil. Potassium moving by more tortuous pathways would have had more opportunity to react chemically with the soil and so become adsorbed into the soil exchangeable K^+ pool. Finally, the majority of added urine may have moved by micropore flow and simply caused piston displacement of the native soil water. In the extreme case, only the native soil water would appear in the first leachates. Since the leachate volume was measured once (after three days), the relative rates of immediate preferential flow could not be determined between the different soil types.

Apart from the physical differences that can occur between these soil types, there are obvious chemical differences which may explain the variation in the amounts of K retained by these soils. For example, the Egmont soil retained the largest amount of K in the soil (Figure 5.2; Table 5.4). This was possibly due to the higher CEC of this soil and its high content of pH dependent charge (which increases with increased soil pH as evidenced in Chapter 4; section 4.3.1). The Tokomaru soil contained the clay mineral vermiculite and so was expected to

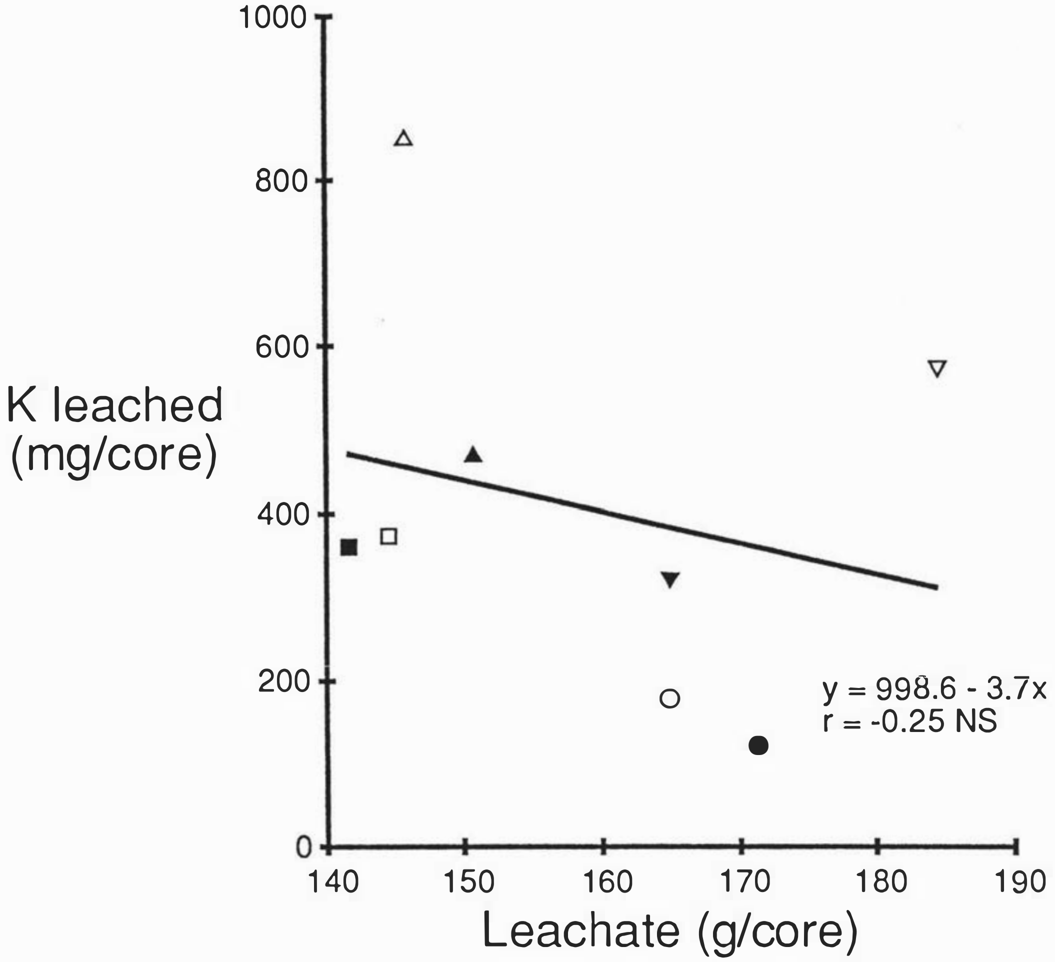


Figure 5.4 The relationship between the amount of leachate collected and the amount of K collected in the leachate during a three day period following an application of dairy cow urine to intact cores of four different soil types. The data points represent the mean values for each soil type (Tokomaru (▽), Manawatu (△), Egmont (○), Pukepuke (□)) and urine treatment (low urine (closed symbols), high urine (open symbols)). Values are means of six replicates.

retain K. However, the Tokomaru soil does not appear to have retained greater amounts of K than the other soils (Figure 5.2; Table 5.4). Similarly, the Pukepuke soil, because of its clay mineralogy, was expected to retain relatively small amounts of urine K, but it does not appear to have retained any more or less than the other soils (Figure 5.2; Table 5.4).

Losses of K due to preferential flow of urine through the soil profile following a urination event have not been identified or discussed by other researchers. There is, however, some evidence which suggests that preferential flow may occur in the data published by Ball et al. (1979) and Quin (1977). This data shows that there were high amounts of NH_4^+ in soil samples collected down to soil depths of 45 cm after an application of urine to soil. These high amounts can have originated only from the urine applications.

5.3.3 The Leaching of Potassium During the Simulated Rain Events

The simulated rain events caused more K to be leached from the cores receiving the more concentrated urine, and considerably more K was leached from all urine treated cores than from the cores that had received only water (Table 5.4). However, when the amount of K leached was expressed as the percentage of applied K (using the formula (K in leachate from urine treated soil - K in leachate from water treated soil) x 100 / K applied in urine) there were no differences between the low and high urine treatments (Figure 5.2). All soils leached similar proportions of the applied K, except for the Pukepuke high urine treatment, which leached significantly more of the applied K than the other soils, particularly from the high urine treatment (Table 5.4).

The amounts of K leached from each simulated rain event are shown in Figure 5.5. For the Tokomaru, Manawatu and Egmont soils there were relatively similar amounts of K leached during each event. This applied to both the high and low urine treatments. In contrast, for the Pukepuke soil the amount of K leached decreased with time.

Irrespective of soil type, only a small proportion (<6%) of the applied urine K was leached during the 90 mm of simulated rainfall (Figure 5.2). These losses are similar to the 7% reported by Saunders and Metson (1959) and the 1-4% reported by Hogg (1968; 1981). However, these latter experiments were of a longer duration and involved greater amounts of rainfall (Table 2.6).

The losses of K reported in this intact core experiment and those of Saunders and Metson (1959) and Hogg (1968; 1981), which involved either intact soil cores or field lysimeters, are much lower than those reported from laboratory experiments using sieved soil

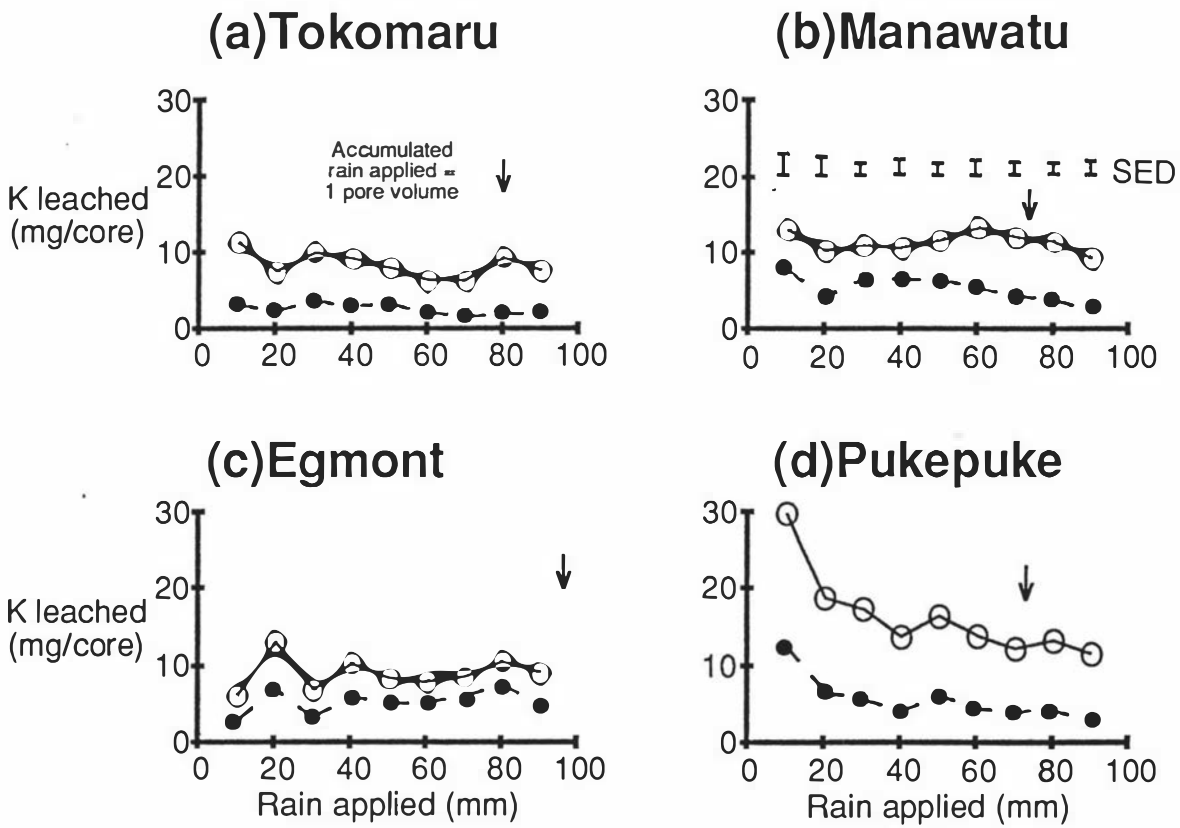


Figure 5.5 The amounts of K leached during each rain event from cores which received the low (●) and high (○) urine treatments. SEDs for all soil types and urine concentrations are presented in Figure 5.5b. Values are means of six replicates.

repacked into columns (Table 2.6). The lower losses from intact cores indicates that much of the applied water moved preferentially through the soil macropores and so by-passed the majority of the urine K retained in the soil micropores and adsorbed on soil surfaces. Very little preferential flow of applied water would be expected in repacked columns of sieved soil. The urine K retained in soil water in repacked columns is therefore likely to be displaced by the incoming rainfall thus leading to high apparent leaching.

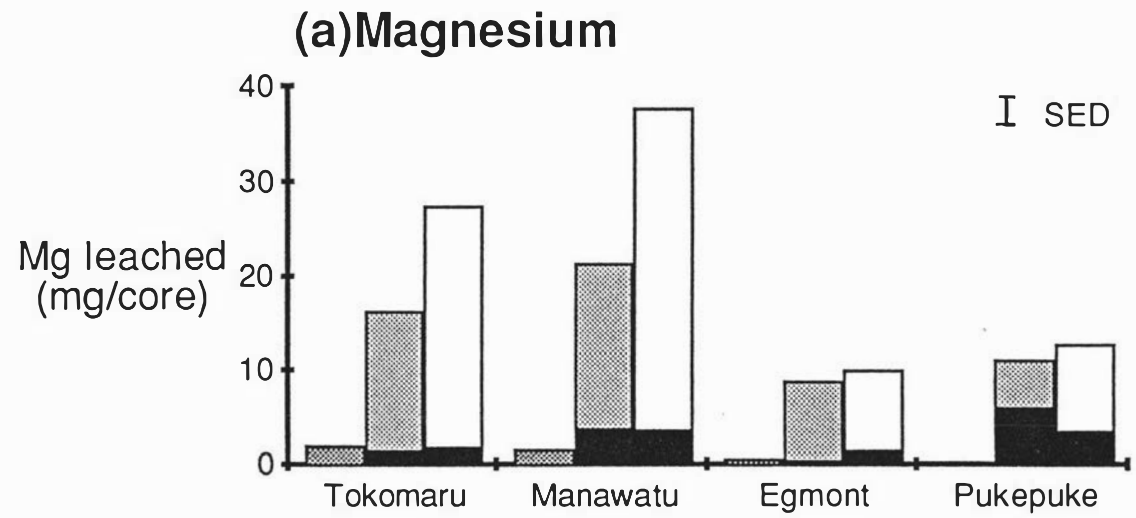
5.3.4 **Leaching of Other Ions**

5.3.4.1 *Ammonium and chloride*

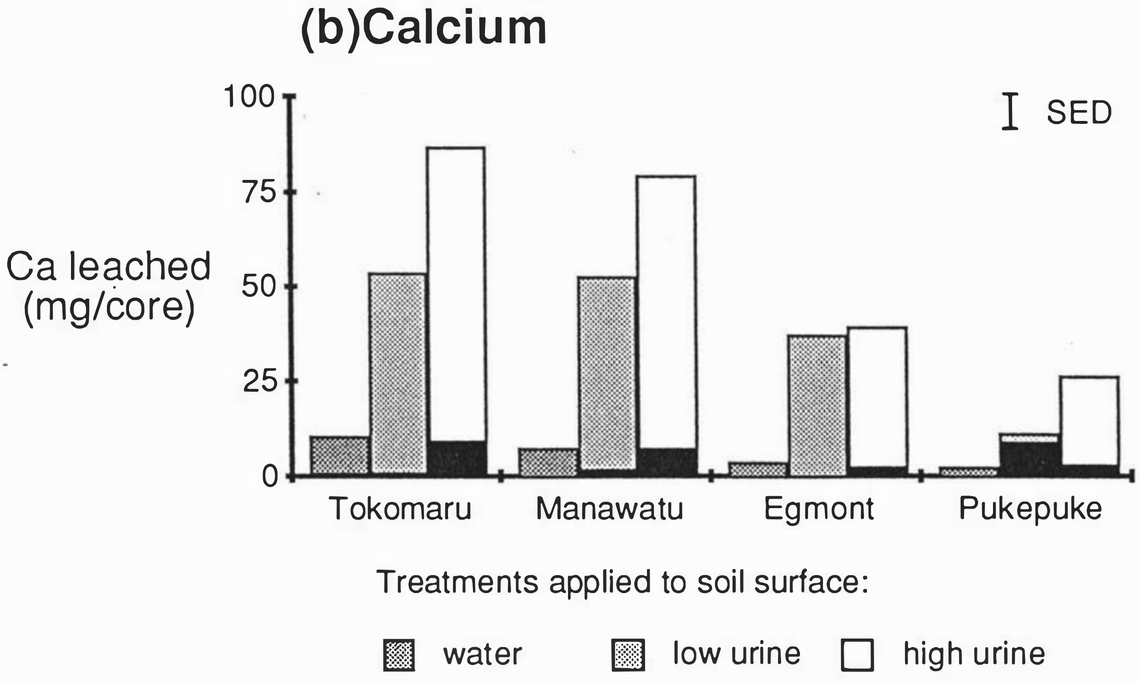
High concentrations of NH_4^+-N (180-4500 μg ml^{-1}) were detected in the leachate collected three days after the urine application. These NH_4^+-N concentrations were much higher than the concentrations measured in the leachate from the cores treated with water (<10 μg ml^{-1}) indicating that most of the NH_4^+ in the leachate, like most of the K, originated from the urine application. Similarly, the Cl^- concentration in the leachates from cores receiving the urine applications were higher (100-750 μg ml^{-1}) than in the leachates from the cores treated with water (<10 μg ml^{-1}) and so must also have originated from the urine application. These high NH_4^+-N and Cl^- concentrations in the leachate from the urine application provide further evidence for preferential flow of urine through the soil cores.

5.3.4.2 *Magnesium, calcium and sodium*

The application of dairy cow urine contributed either 15 or 38 mg Mg to soil cores receiving the low and high urine concentrations, respectively. The amounts of Mg applied in the urine were similar to the total amounts of Mg leached from the Tokomaru and Manawatu soils (Figure 5.6). Less Mg was leached from the Egmont and Pukepuke soils than was applied in the treatments (Figure 5.6). Urine contains very little Ca (Table 5.3), however up to 90 mg Ca per core were leached during the experiment (Figure 5.6). Greater amounts of Ca (and Mg) were leached from the Tokomaru and Manawatu soils reflecting the higher native Ca (and Mg) contents in these soils (Table 5.5). More Ca was leached immediately following the urine treatments, especially the high urine treatment, reflecting the displacement of Ca^{++} from the soil surface cation exchange sites by K^+ and NH_4^+-N from the urine. The difference in the amounts of Ca leached from the two urine treatments was not as apparent with the Egmont soil (Figure 5.6), presumably because this soil has a high CEC and increased cation adsorption occurs after an application of urine due to increased soil pH (section 4.3.1). The significance of the quantities of Ca and Mg lost from urine spots per annum is discussed in section 8.5.1.3.



(a)Magnesium



(b)Calcium

Figure 5.6 The sum of the amounts of Mg and Ca in the leachates collected immediatly after the urine application (■) and the rain events for each soil type. Values are means of six replicates.

Table 5.5 The sum of the water soluble and the exchangeable cations in all four soils before and after the experiment for soil cores receiving urine of low and high concentrations (0-15 cm depth of soil; me 100 g^{-1}).

	Tokomaru	Manawatu	Egmont	Pukepuke	SED
Before treatment application					
Potassium	0.23	0.55	0.25	0.15	0.043
Calcium	6.61	7.00	4.30	4.76	0.354
Magnesium	1.16	1.15	0.95	0.99	0.086
Sodium	0.04	0.05	0.04	0.04	0.006
At end of experiment					
Low urine					
Potassium	0.92	0.97	1.56	0.85	0.146
Calcium	5.57	6.00	4.13	3.92	0.640
Magnesium	0.93	1.31	0.85	0.82	0.149
Sodium	0.05	0.06	0.09	0.07	0.022
High urine					
Potassium	0.58	0.81	0.80	0.37	0.146
Calcium	5.93	6.38	4.22	4.37	0.640
Magnesium	0.99	1.35	0.77	1.12	0.149
Sodium	0.04	0.06	0.08	0.09	0.022

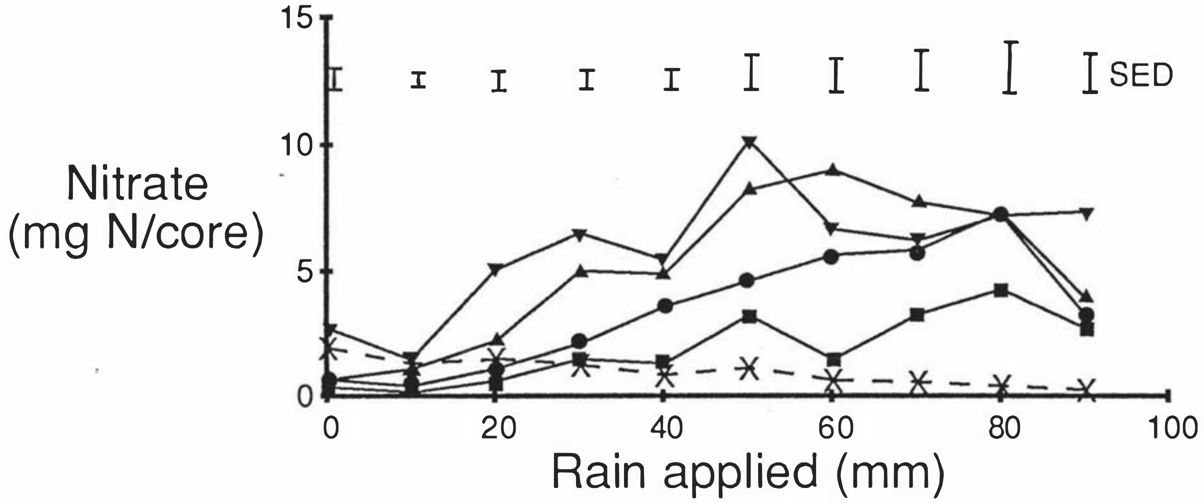
The results shown in Figure 5.6 agree with the findings of Hogg (1981) and Steele et al. (1984) that leaching losses of Ca and Mg can occur from urine affected soil. These authors showed that Ca, Mg and K were leached in association with NO_3^- ions. In this experiment with intact soil cores, nitrification of the NH_4^+ (derived from the urine) to NO_3^- began when 20 mm of rain had been applied (six days after treatment application; Figure 5.7). The nitrification rate in the Egmont and Pukepuke soils that received the high urine concentration appears to have been inhibited (Figure 5.7). The resulting high soil pH and the lower concentrations of NO_3^- produced may be reasons for the lower Ca and Mg leaching losses in these soils (Figure 5.6). Slow nitrification in the Egmont soil was also observed in the laboratory incubation study described in Chapter 4 (section 4.3.1; Figure 4.5) and again in the slow draining cores of the next experiment with intact soil cores (Chapter 6; section 6.3.4; Figure 4.6).

The concentration of Na^+ was measured in some of the leachates to determine if any Na was being leached from these cores. The concentrations were very low (<5 μg ml^{-1}) and no further Na^+ analyses were carried out on the leachates.

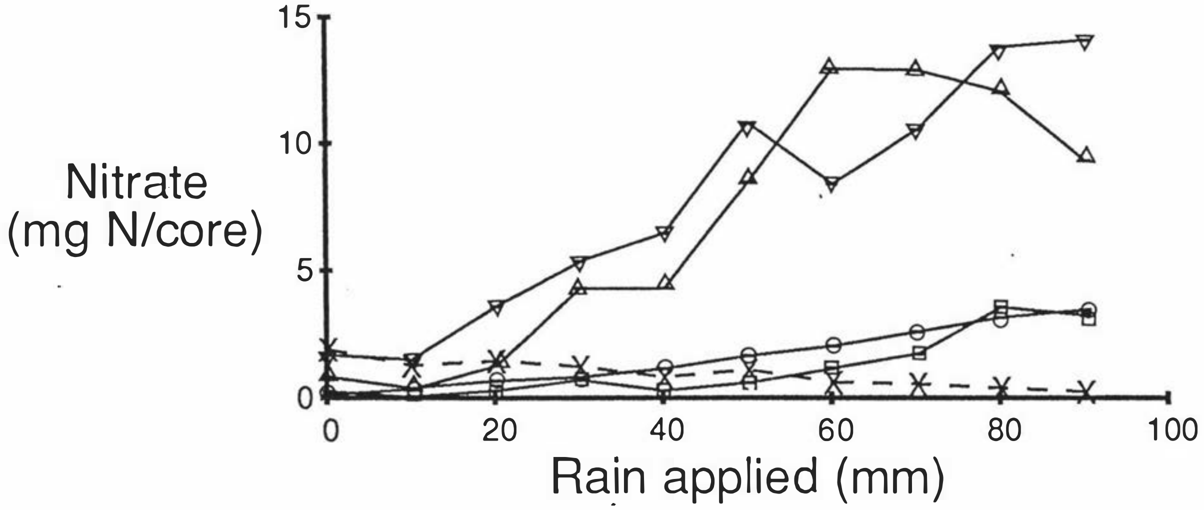
5.3.4.3 *Cation anion balance in the leachates*

The sum of the charge due to major cations (K^+, NH_4^+, Ca^{++} and Mg^{++}) and the sum due to anions (NO_3^- and Cl^-) in the leachates collected from each rain event are shown in Figure 5.8. Data used to calculate these sums are given in Figures 5.4, 5.6 and Appendix II. For four of the experimental treatments, the equivalent charges on the cations provided an approximate balance for those of NO_3^- and Cl^-. This indicated that NO_3^- and Cl^- were the main anions leached from these cores. However, in the leachates from some cores (notably in leachates from the Pukepuke high urine treatment; Figure 5.8h) the major cation charge exceeded the sum of the NO_3^- and Cl^- charge. This difference between the ion charge balance of different soil cores may be due to two factors:

Firstly, the amount of NH_4^+ leached from the soil during the initial rain events may have been overestimated. These leachates may have contained urine-N in the form of urea which could have hydrolysed to NH_4^+ as the leachate stood for three days while the soil drained into the collection container. This would generate NH_4^+ ions (which could contribute to the sum of cations), and bicarbonate (HCO_3^-) or OH^- ions (which were not initially included in the sum of anions). Evidence for this is provided by the high NH_4^+ content in the leachates from the first 10 mm of rain (Appendix II) of the treatments which had the largest cation/anion imbalance (e.g., Pukepuke low and high urine treatments). Unhydrolysed urea in the leachates was detected in the experiment described in Chapter 6 (section 6.3.2.3; Table 6.5). Furthermore,



(a)Low urine



(b)High urine

Figure 5.7

The amounts of nitrate leached from cores of different soil types (Tokomaru (▽), Manawatu (△), Egmont (○) and Pukepuke (□)) and urine treatments (water (✗), low urine (closed symbols), high urine (open symbols)). SEDs for all soil types and urine concentrations are presented in Figure 5.7a. Values are means of six replicates.

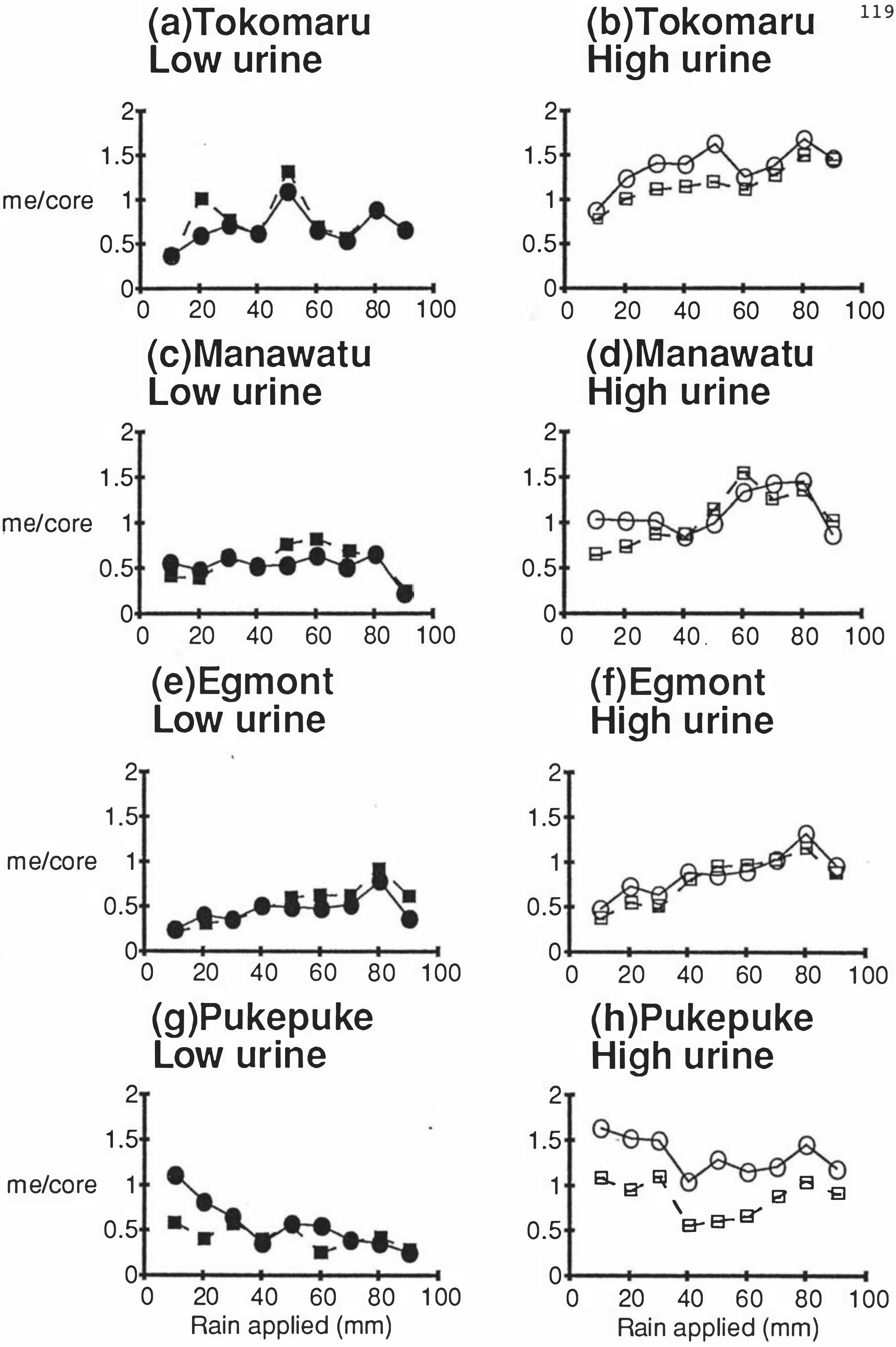


Figure 5.8 Sum of cations ($\Sigma(K^+, NH_4^+, Mg^{++}, Ca^{++})$;○) and anions ($\Sigma(NO_3^-, Cl^-)$;□) collected in the leachate from each rain event. Values are means of six replicates.

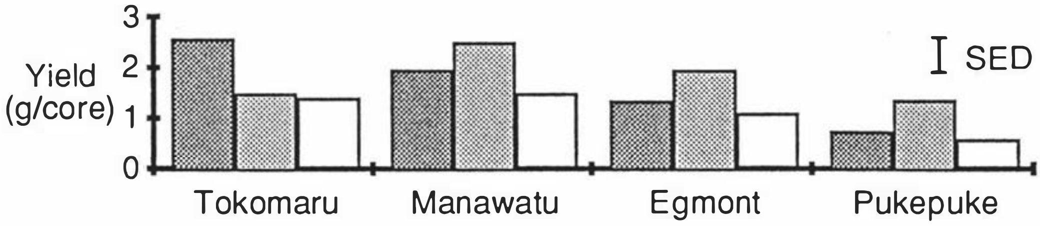
the pHs of selected leachates (i.e., where there was a large difference between the amount of anions and cations present in the leachate) were in the range of 8.2 to 8.6, which indicates that there were significant amounts of HCO_3^- or OH^- ions present. The HCO_3^- ions may have originated from the hydrolysis of urea either in the collected leachate as described above, or in the soil. Selected leachates were titrated to pH 7 with a weak acid (0.005M hydrochloric acid) which indicated that HCO_3^-, OH^- and dissociated organic acids may have contributed between 0.1 and 0.5 me core^{-1} to the anion total.

Secondly, there may have been sulphate ($SO_4^=$) present in the leachate which was not included in the sum of anions in Figure 5.8. Boswell (1983) has measured leaching losses of 54 kg S ha^{-1} per 28 mm of drainage from urine affected soil in an experiment with columns of repacked sieved soil. Measurement of the concentration of $SO_4^=$ in the selected leachates showed that up to 0.1 me S was leached per core. This is equivalent to 1 kg S ha^{-1} per 4.5 mm of drainage which is considerably less than that reported by Boswell (1983).

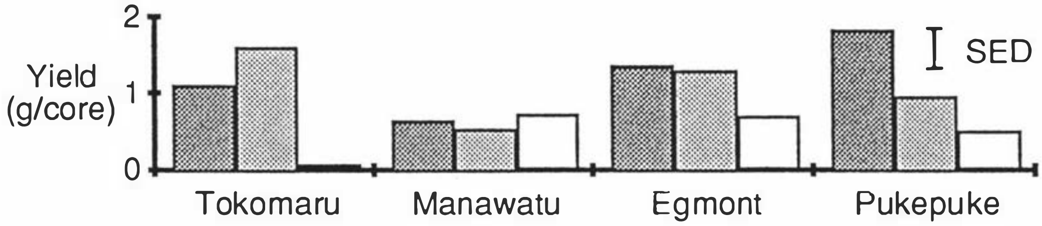
Where the cation charge was not balanced by the NO_3^- plus Cl^- charge (Figure 5.8) the difference between cations and anions was never greater than 0.6 me core^{-1}. The results from measurements of the selected leachates suggest that the charge imbalance could be accounted for by HCO_3^-, OH^-, dissociated organic acids and $SO_4^=$. In general, the major ions that were lost by leaching from the simulated rain events were (in order of the me core^{-1} lost) $NH_4^+ > Cl^- > K^+ > Ca^{++} > NO_3^- > Mg^{++}$. If the relative concentrations of these ions in the applied urine is taken into consideration then the soil can be expected to show a net gain in K^+ and NH_4^+ (and NO_3^- through nitrification) and a net loss of Ca^{++}, while the Mg^{++} status of the soil will remain relatively constant. This aspect is considered further in section 5.3.8 where the amounts of exchangeable cations present in the soil before and after the experiment have been measured.

5.3.5 Herbage Yield

The yields of clover, grass and the sum of both are shown in Figure 5.9. Whereas, the yields from the cores receiving water and urine of low concentration were similar, herbage yields (especially the grass yield) of the high urine treatment were reduced (Figure 5.9). Seven days after application of the high urine treatment, the herbage turned brown around the leaf margins and some of the plants died. This effect has been referred to as urine scorch or urine burn and has been observed in the field (Doak, 1952; Holmes, 1968; Richards and Wolton, 1975), particularly where applications of highly concentrated urine occur (e.g., concentration of urine N >1%; Quin, 1977). The main effect of urine burn is thought to be on (a)Clover yield



(b)Grass yield



(c)Total yield

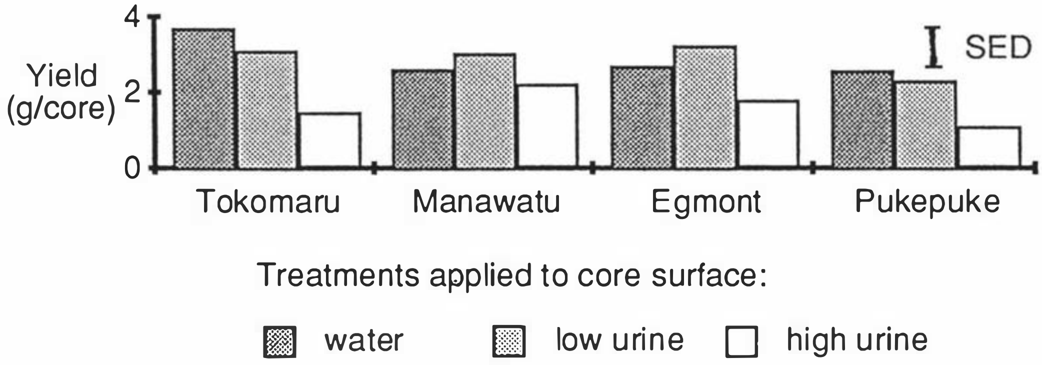


Figure 5.9 The dry matter yield of clover, grass and total herbage on cores of the four different soil types. Values are means of six replicates.

the root system of the plants rather than on the leaves (Richards and Wolton, 1975), and is probably caused by a combination of high salt and ammonia concentrations, high alkalinity and the presence of a hormonal inhibitor in the urine (Doak, 1952; Russell, 1973; Bernstein, 1975).

Despite the fact that pasture growth can be suppressed initially by urine burn, significant increases in pasture yield can occur later over time periods of at least 3 months after both cow and sheep urinations (Sears et al., 1948; Sears, 1953; Wolton, 1955; Watkin, 1957; Norman and Green, 1958; Saunders and Metson, 1959; Dale, 1961; Mundy, 1961; Herriott and Wells, 1963; Lotero et al., 1966; Carran et al., 1982; Ledgard et al., 1982; Ledgard and Saunders, 1982; Saunders, 1984). Most of the observed increases in herbage yield have been associated with changes in pasture composition. Usually an increase in the grass content and a decrease in the clover content occurs. The increase in grass species content is regarded as a response to the nitrogen in the urine. In this experiment, increased grass yields and decreased clover yields relative to the water treatment were not obvious even in the low urine treatment (which was not visually affected by urine burn; Figure 5.9). A possible reason for this lack of response is that the time period of 30 days was too short and the temperature too low to allow for the pasture to respond to the urine application. Dale (1961), while observing the effect of sheep urine on pasture, found that in some cases pasture plant growth did not increase until 10 months had elapsed since the time of urine application.

5.3.6 Uptake of Potassium By the Herbage

The concentration of K^+ in the herbage and the amount of K recovered by the herbage are presented in Figure 5.10. The K^+ concentrations in both the clover and grass on cores receiving the water treatment were above 2.0% and 2.2%, respectively. Such K concentrations have been suggested by McNaught (1970) as being optimal for growth. The one sub-optimal exception was the water treated Pukepuke soil, which had K^+ concentrations of 1.5 and 1.8% for the grass and clover, respectively. These low herbage K^+ concentrations suggest that the Pukepuke soil may have responded more to the application of urine compared with other soils. Such a result was not apparent (Figure 5.9).

Generally, the amount of K taken up by the herbage was similar in the Tokomaru, Manawatu and Egmont soils but lower in the Pukepuke soil (Figure 5.10c). The lower K recovery by herbage grown on the latter soil can be attributed mainly to the lower clover dry matter yield (Figure 5.8a) and to a lower herbage K^+ concentration, particularly in the grass (Figure 5.9b).

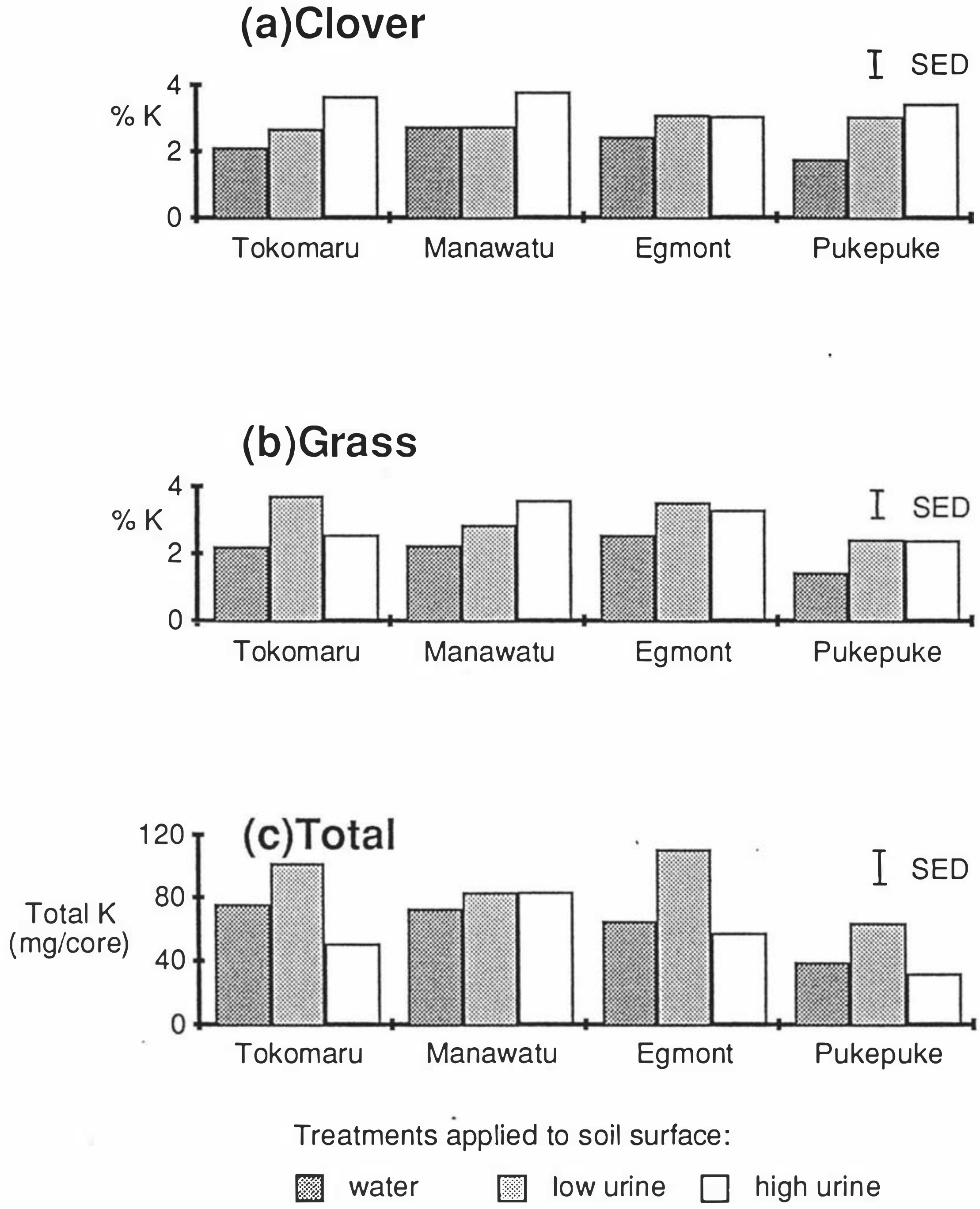


Figure 5.10 The K^+ concentration in the herbage (clover and grass) and the total amount of K recovered in the herbage from cores of the four different soil types. Values are means of six replicates.

For all soils except the Manawatu, the lowest total K uptake by the herbage occurred when the high urine concentration was applied (Figure 5.10c), the cause being the reduced herbage yields (Figure 5.9) as previously discussed (section 5.3.5).

The amount of K taken up by the herbage grown on urine treated soil amounted to 1-9% of the total K applied (Figure 5.2). The percentage recovery of urine K in the herbage growing on the low urine treatments was higher (5-9%) than in herbage growing on the high urine treatment (1-3%). The results for K recovery in the herbage growing on urine affected Tokomaru soil are similar to those collected in the field experiment (Chapter 3), which showed that 6% of the K applied in urine (of a similar K^+ concentration to the low urine treatment used in this experiment) was recovered by the herbage during the first five weeks after application (Figure 3.3).

5.3.7 Uptake of Other Cations By the Herbage

The cations Ca^{++}, Mg^{++} and Na^+ are present in urine in small amounts (Table 5.3) and so urine does not contribute directly to increased concentrations of herbage Ca^{++}, Mg^{++} and Na^+. However, the concentration of these cations in pasture plants can be reduced by applications of K as fertiliser or urine (e.g., McNaught et al., 1973; Metson and Saunders, 1978; Smith et al., 1983; Barber, 1984; G.S. Smith et al., 1985). A high K/(Ca + Mg) ratio in the herbage (Metson et al., 1966) can lead to metabolic problems particularly hypomagnesaemic grass tetany in dairy cows. Similarly, a high K/Na ratio in the herbage ingested by lactating dairy cows can lead to a Na deficiency (Towers, 1983; Towers and Smith, 1983).

The effect of urine on the concentration of Ca^{++}, Mg^{++} and Na^+ in the herbage of this experiment was extremely variable with no consistent trends resulting from the different treatments (Appendix III). However, the application of urine significantly increased the K/(Ca + Mg) ratio in the herbage of all the soils except the Manawatu soil (Table 5.6). Saunders (1984) also found similar increases (from 0.9-1.5 to 2.0-2.7) in the K/(Ca+Mg) ratios in herbage affected by dung and urine. He concluded that if the grazing pressure is high enough to ensure that stock eat significant quantities of pasture growing in areas affected by dung and urine then metabolic disorders may occur.

Table 5.6 The mean K/(Ca+Mg) ratios in the herbage (sum of clover and grass fractions).

Soil type	Treatment Water	Low urine	High urine
Tokomaru	0.79	1.07	1.38
Manawatu	1.56	1.16	1.66
Egmont	1.45	2.12	2.74
Pukepuke	0.77	1.49	1.96
SED		0.326	
significance of:			
soil main effect		***	
urine main effect		***	

5.3.8 Soil Analysis

At the end of the experiment, between 29 and 50% of the applied urine K was still present in the soil (Figure 5.2). This K was mainly in the form of exchangeable K^+ rather than water soluble K^+ (Figure 5.11). Amounts of K remaining in soil as exchangeable K^+ after applications of dairy cow urine were smaller, i.e., 0 to 33% of the K in other studies (Saunders and Metson, 1959; Davies and Hogg, 1960; Hogg, 1968; 1981), than those reported in this controlled-climate chamber experiment. However, all of these trials were conducted over a much longer time interval (i.e., between 6 months and 4 years) than the one month used in this experiment. The results from the experiment in Chapter 3 showed that the plant uptake of urine K from a field soil continued over a 5 month period.

The possible retention of urine K by the soil in a nonexchangeable form has already been discussed (section 5.3.1). During and McNaught (1961) and Hogg (1968) have also suggested that some of the K unaccounted for in their experiments may have been converted into a nonexchangeable form. The proportion of urine K that is retained as nonexchangeable K^+ may not be as high as when other sources of K input are used (e.g., KCl). This is because the large amounts of NH_4^+ in the urine may compete with K for cation exchange sites on the soil surface (Wolton, 1955). If both K and NH_4^+ are applied together there can then be a greater retention of the NH_4^+ compared with the K (Joffe and Levine, 1947) because NH_4^+ competes strongly for the clay-interlayer adsorption sites (Nommik, 1965). However, the results of the incubation experiment conducted over 30 days (section 5.2.8) failed to indicate any difference between urine and KCl in the amount of K that was though to be retained as nonexchangeable K^+ (Figure 5.3).

The greatest increase in soil K (water soluble and exchangeable K^+) occurred in the 0-7.5 cm depth of soil with a smaller increase in the 7.5-15 cm depth (Figure 5.11). These results are consistent with those of Hogg (1981) who, after a urine application, measured an increase in exchangeable soil K^+ mainly in the 0-7.5 cm depth of soil and to a lesser extent at depths down to 90 cm. In a field study, under dairy cattle grazing, Saunders (1984) found that there was an increase in soil exchangeable K^+ in excreta affected areas at a soil depth of 0-7.5 cm, but at the 7.5-15 cm depth differences were small and inconsistent.

Significantly different amounts of urine K were retained in the different soils ($P<0.05$; Table 5.4). In increasing order of K retention the soils were Manawatu 29 and 32%, Tokomaru 41 and 42%, Pukepuke 48 and 31%, and Egmont 50 and 48%, for the high and low urine treatments, respectively (Figure 5.2). Differences between the soils in the amount of K

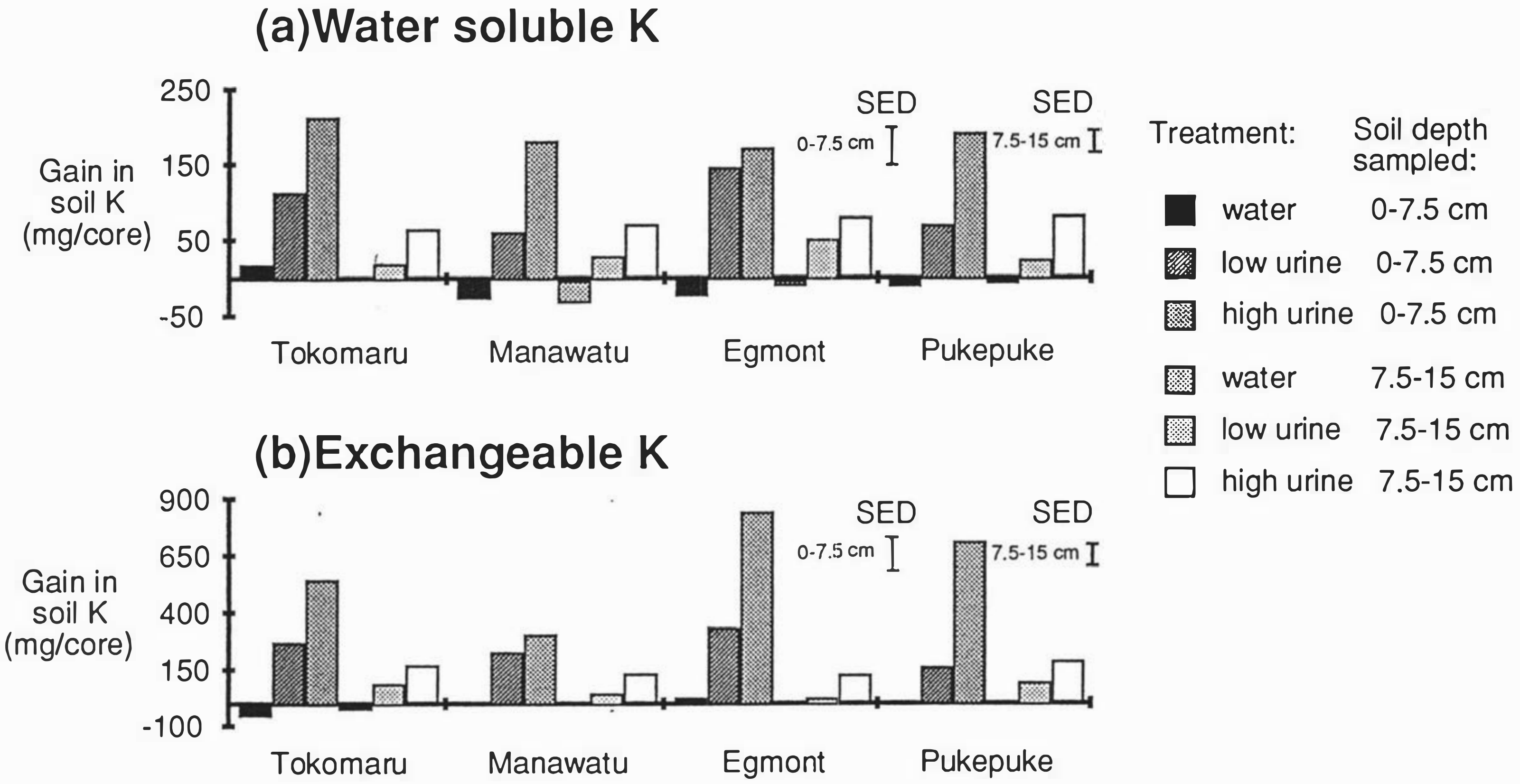


Figure 5.11 The gain in the amount of water soluble and exchangeable K^+ in the four soils, expressed as the difference in amount of K before and after the experiment. Values are means of six replicates.

retained in the soil could possibly be attributed to differences in soil chemical properties, particularly CEC and base saturation, and soil physical properties, particularly those relating to the pathway taken by urine and subsequent rain events through the soil core (section 5.3.2.1).

As predicted in section 5.3.4.3, the increase in the amounts of exchangeable K^+ in the urine treated soils was accompanied by a decrease in exchangeable Ca^{++}, particularly on the Tokomaru and Manawatu soils which have high native Ca contents (Table 5.5). This Ca (which represented between 0.2 and 1% of the total exchangeable Ca^{++} initially present in the soil cores) appeared mainly in the leachate from the rain events (Figure 5.6; Appendix II). The differences in the exchangeable Mg^{++} and Na^+ content of the soil between the beginning and end of the experiment were not statistically significant (Table 5.5). As previously discussed, the amount of Mg applied in urine was similar to that leached during the experiment and the amount of Na leached was very small, hence the soil Mg and Na contents remained similar throughout the experiment.

No measurements of soil exchangeable NH_4^+ were carried out at the end of the experiment. However, the soil pHs at the end of the experiment (Table 5.7) were higher, especially in the high urine treatment, than at the beginning (Table 5.1). This indicated that nitrification of the urea applied in the urine treatments was not complete. It is therefore likely that there was some urine derived NH_4^+ present on the soil cation exchange sites at the end of the experiment, particularly in the high urine treatment.

5.4 CONCLUSIONS

Analysis of the fate of urine K applied to naturally structured intact soil cores, with moisture contents resembling field capacity, showed that over a 30 day time period after urine deposition, a large proportion (29-50%) of the urine K remained in the soil, mainly as exchangeable K^+. Significant losses of urine K occurred during the first three days after urine application, when 5-39% of the urine K was collected in the leachate. Given the high concentration of K^+ (and NH_4^+ and Cl^-) in the leachate which was collected three days after urine application, and the small volume of liquid (equivalent to 0.11-0.15 pore volumes) applied, it is assumed that some urine flowed preferentially through the soil cores via a network of soil macropores.

Differences existed in the amounts of urine K retained by each soil, with Egmont retaining 50 and 48%, Pukepuke 48 and 31%, Tokomaru 41 and 42%, and Manawatu 29 and 32% of the K applied as urine of high and low concentration, respectively. Whether these

Table 5.7 The mean soil pH at the end of the experiment (0-7.5 cm depth).

		Treatment		
Soil type		Water	Low urine	High urine
Tokomaru		6.2	6.1	7.2
	SEM	0.24	0.14	0.36
Manawatu		6.2	6.3	6.6
	SEM	0.05	0.11	0.28
Egmont		6.0	5.7	6.7
	SEM	0.08	0.07	0.34
Pukepuke		6.2	6.3	6.8
	SEM	0.09	0.21	0.38

differences were due to diverse soil chemical properties and/or different pathways for K movement through each soil could not be determined from this experiment, but this aspect will be examined in more detail in Chapter 6.

The results from this controlled-climate chamber experiment have shown that a small proportion (<6%) of urine K was leached during simulated rain events over a 30 day period. This information should not be extrapolated in order to predict the amount of urine K that will be leached on an annual basis. This aspect will also be studied in Chapter 6.

The recovery of the urine K by plant uptake accounted for less than 10% of the K applied. The time interval of this experiment (30 days) is similar to the interval used to spell paddocks between grazings on a dairy farm. Therefore the results from this experiment, and the field experiment (Figure 3.3), suggest that little of the urine K would be recycled to the above ground portion of the K cycle during the time interval between grazing events. Instead the return of urine K to the above ground portion of the cycle occurs over a longer time scale of one growing season (section 3.3.3.2).

The application of dairy cow urine to soil also influences the fate of ions native to the soil. Between 0.2 and 1% of the initial amounts of exchangeable Ca^{++} present in the soil were leached during the experiment, after displacement from the soil surface cation exchange sites by the K^+ and NH_4^+ ions originating from the urine. The result of the excessive urine K inputs into the soil and the enhanced leaching of Ca was that the ratio of K/(Ca + Mg) in the herbage increased. The high ratio may lead to metabolic disorders in dairy cows if urine affected herbage is a major component of their diet.

The main cations leached from the rain events were NH_4^+, K^+ and Ca^{++}. The anions associated with these were mostly NO_3^- and Cl^-. Significant amounts of N were leached from the urine treated soil following both the urine application and the simulated rain events. This N was measured as NH_4^+ and NO_3^- but immediately following the urine application there may also have been some N leached as urea. In Chapter 6 this aspect of N leaching will be investigated further.

Irrespective of the K concentration, the relative proportions of applied urine K that were recovered in the leachate, soil and herbage were similar. The high concentration urine also suppressed herbage growth and nitrification, particularly in the Egmont and Pukepuke soils.

CHAPTER 6

PREFERENTIAL FLOW OF DAIRY COW URINE AND RAINFALL THROUGH INTACT SOIL CORES

6.1 INTRODUCTION

The experiment conducted in Chapter 5 demonstrated that the greatest loss of K from dairy cow urine applied to intact soil cores resulted from leaching which occurred during the first three days after urine deposition. This loss was thought to have resulted mainly from the preferential flow of urine through the network of macropores in the soil. The evidence for this was that the small volume of urine applied represented only 0.11-0.15 pore volumes of a core, which would have been too low for the urine front to reach the bottom of the core by piston-type displacement of the initial soil water. In addition, the concentration of K^+ in the leachate was similar to that of the applied urine, suggesting that some urine had passed directly through the core.

The amounts of K that were leached immediately after urine deposition were different for each of the four soil types used in the experiment in Chapter 5 (section 5.3.2.1). However, because the drainage from the urine deposition was collected over a three day time period it was not possible to identify whether these differences were due to the differences in the physical or the chemical properties of each soil. The experiment reported in this chapter was designed to investigate the period when urine flowed through the soil cores. By labelling urine with tritiated water (^3H_2O) and comparing the breakthrough curves of the relative activities of tritium and the relative concentrations of K^+ in the leachates, it was hoped to be able to identify the relative influence that soil physical and chemical properties have on urine K movement through different soil types. The breakthrough pattern of tritiated water should be unaffected by ion exchange processes with soil surfaces and reflect the physical constraints placed on the flow of water through the core. Depending on flow rate, ion exchange processes may or may not significantly influence the passage of K^+ ions through the soil.

In comparison to the amount of K leached from intact cores during the three days after the urine application, the amount of K leached from subsequent simulated rain events was very small (<6% of the applied K; section 5.3.3). In this previous experiment, because leachate was collected three days after each rain event, it was not possible to differentiate the proportions of leachate derived by macropore and micropore flow. It is likely that K^+ ions moving by macropore flow are influenced less by soil chemical properties (e.g., cation exchange) than K^+ions moving in slower micropore flow. By labelling the water of a simulated rain event with tritium it was hoped that the extent to which simulated rain water moved preferentially through the soil core, and what effect this had on K leaching, could also be established. The experiment described in Chapter 5 was carried out over a 30 day period and involved the application of 10 simulated rain events. Whether more rainfall events would have increased the amount of K that was leached is unknown. However, such information is important for calculating the annual amount of K that is lost from grazed pasture as this influences K fertiliser requirements.

The experiments described in this chapter utilise tritiated urine and tritiated simulated rain to study the influence of preferential flow of urine and rain water through soil on K leaching. As in the previous study (Chapter 5) intact soil cores were used and the experiment was conducted in a controlled-climate chamber.

6.1.1 Objectives

The main objectives of this study were to:

i. measure the proportion of urine K that moved preferentially through intact cores immediately following deposition.

ii. measure the proportion of flow that moved preferentially following a simulated rain event.

iii. predict the amount of K leached on an annual basis from a urine deposition to pasture soils.

6.2 MATERIALS AND METHODS

6.2.1 Collection and Preparation of Soil Cores

Seven intact soil cores (15 cm diameter x 15 cm depth) were collected from the four soil types as described in section 5.2.1.1. Extra soil from around each core was also collected, and the water soluble and exchangeable K^+ and Ca^{++} content of each sample was determined (section 5.2.6.2). For each soil, four intact cores of similar initial exchangeable K^+ contents were selected.

Each core was prepared for the experiment as described in section 5.2.1.2. At the start of the experiment the mean gravimetric water contents of the Tokomaru, Pukepuke, Egmont and Manawatu cores were 0.38, 0.45, 0.53 and 0.30 g g^{-1} respectively.

6.2.2 Urine Application

The dairy cow urine used in this experiment was collected from the Massey University No. 3 dairy farm at a morning milking in August 1986. The urine was frozen immediately and stored until required. Sixteen hours before the experiment began 3.6 dm^3 of the urine were removed from the freezer, thawed at room temperature and 0.996 mCi of tritiated water (2 ml, ^3H_2O) were thoroughly mixed with it. The chemical composition of this urine is given in Table 6.1.

Prior to urine application, the herbage on each core was trimmed to 2 cm above ground level and the core was then placed on a 15 cm plastic funnel supported on a wooden frame. Two hundred ml of the tritiated urine were applied and the leachate collected in 25 ml beakers. The time taken to collect each leachate sample of 25 g (1 g = 1 ml of leachate) was recorded. Each sample was filtered through Whatman No.5 filter paper and frozen with 2 drops of toluene to inhibit microbial growth. At a later date these were thawed overnight at room temperature and the concentrations of K^+, Cl^-, NH_4^+-N, NO_3^--N, kjeldahl-N and Ca^{++} (section 5.2.6.1) and the activities of tritium (section 6.2.7) in each sample were measured. When each core had finished draining it was placed on a 14 cm diameter plastic container inside a larger plastic container (Figure 5.1), then placed in a controlled-climate chamber as described in section 5.2.1.2. The environmental conditions of the chamber were set to simulate a late August climate (section 5.2.1.2). The cores were arranged in a randomised block design within the controlled-climate chamber using soil type as the treatment.

6.2.3 Simulated Rainfall Applications

On every second day following the urine application, 30 mm of simulated rainfall were applied as six events of 5 mm. Each event comprised 90 ml of deionised water applied from a wash bottle; the interval between applications was 1.5 hours. In the Manawatu region drainage can be expected to occur normally during the period May to October. Meteorological data for this period shows that on average rainfall exceeds evapotranspiration by 300 mm (New Zealand Meteorological Service, 1973). Hence the rain events in this experiment continued until 300 mm of drainage had been collected.

Table 6.1 The chemical composition of the urine (methods of analysis are detailed in sections 5.2.6.1 and 6.2.7).

Total Salts	30 mg ml^{-1}
Potassium	6.6 mg ml^{-1}
Total Nitrogen	5.5 mg ml^{-1}
Chloride	2.5 mg ml^{-1}
Calcium	50 μg ml^{-1}
Sulphate-S	360 μg ml^{-1}
Nitrate-N	<0.01 μg ml^{-1}
Phosphate-P	<0.01 μg ml^{-1}
pH	8.6
Tritium	0.277 μCi ml^{-1}

During the first 3 days of rain, the leachates collected in the inner and outer plastic containers were analysed separately as in the previous controlled-climate chamber experiment (section 5.2.1.2). There was no evidence for preferential flow of leachate down the interface between the galvanised steel cylinder and the soil, and for the remainder of the experiment the samples in both containers were combined together for chemical analysis. All leachates were processed, and the concentrations of K^+, Cl^-, NH_4^+-N, NO_3^--N, kjeldahl-N and Ca^{++} and the activities of tritium were determined as described in section 6.2.2.

There was considerable variation in the rate of drainage observed between the cores of similar and different soil types. Seven of the 16 cores had noticeably slower drainage flows than the rest. In these slow cores the 5 mm of rain often ponded on the surface of the soil core for longer than the 1.5 hour interval between rain events. When this happened the rain events were discontinued on these cores until the water had drained away. The increasing weights of these slow draining cores showed that they were becoming progressively wetter with time, indicating that there may have been a breakdown in their porosity due to saturation (Beven and Germann, 1982). It was possible to collect only 140 to 250 mm drainage from these cores rather than the 300 mm that was collected from the faster draining cores.

6.2.4 Final Simulated Rainfall Application

In accordance with the second experimental objective, the final 30 mm of simulated rain were applied as deionised water spiked with 0.277 μCi of tritium per ml. The leachate was collected in 25 g aliquots and the time taken to collect each was recorded. These samples were processed, and the concentrations of K^+, Cl^-, NH_4^+-N, NO_3^--N, kjeldahl-N and Ca^{++} and the activities of tritium were determined as described in section 6.2.2.

6.2.5 Herbage Analysis

After the last leachate had been collected (forty eight hours after the application of the final simulated rain event) the herbage was harvested at ground level. The herbage was then dissected into grass and clover fractions, dried, weighed, digested and the K^+ concentration of the extract was measured as described in section 3.2.4.

6.2.6 Soil Analysis

At the end of the experiment the cores were oven dried at 65^{o}C for 48 hours, after which time the cylinder surrounding the soil core could be removed easily. Each soil core was dissected into 3 sections (0-5, 5-10 and 10-15 cm depths) which were dried at room temperature. After sieving (<2 mm) the soil pH was measured in the three soil depths (section 3.2.5.2). The amounts of water soluble and exchangeable K^{+} and Ca^{++} (section 5.2.6.2) and exchangeable NH_4^{+}-N (section 6.2.8) were determined in each depth.

6.2.7 Tritium Analysis in Leachate

One ml of leachate was placed in a scintillation vial with 10 ml of scintillation cocktail. The cocktail was prepared from 4 g of PPO (2,5-Diphenyloxazole) plus 0.1 g of POPOP (1,4-Di-(2-(4-methyl-5-phenyloxazolyl))-benzene) dissolved in 670 ml of toluene and 330 ml of triton-X 100 (Patterson and Greene, 1965).

The radioactivity of the samples was measured by a Beckman LS 3801 scintillation counter. Recorded light emissions (counts per minute) were corrected for chemical and colour quenching by using a set of quench standards prepared to generate a relationship between H-number (Beckman LS 3801 Liquid Scintillation Systems Operating Manual) and counting efficiency (a quench curve; Figure 6.1). The quench curve was derived from a series of vials containing equal activities of tritium but varying ionic strengths and colour due to the addition of varying amounts of leachate collected from the urine application in the previous experiment. Initially a separate quench curve was prepared for each soil type but there were no differences between these curves. The mean curve was used to correct counts per minute data to disintegrations per minute.

6.2.8 Analysis of Exchangeable Ammonium in Soil

Two g of dried, sieved (<2 mm) soil were shaken on an end-over-end shaker with 20 ml of 2M potassium chloride in 50 ml polypropylene, screw-capped centrifuge tubes for 2 hours (Adamsen et al., 1985). After shaking the tubes were centrifuged at 10,000 rpm for 5 minutes on a Sorvall RC5C centrifuge. The supernatant was collected after filtering through Whatman No. 5 filter paper and was frozen immediately, prior to storage. The NH_4^{+}-N concentrations of the thawed samples were determined by the continuous flow, bubble segmented autoanalyser method of Technicon (1976).

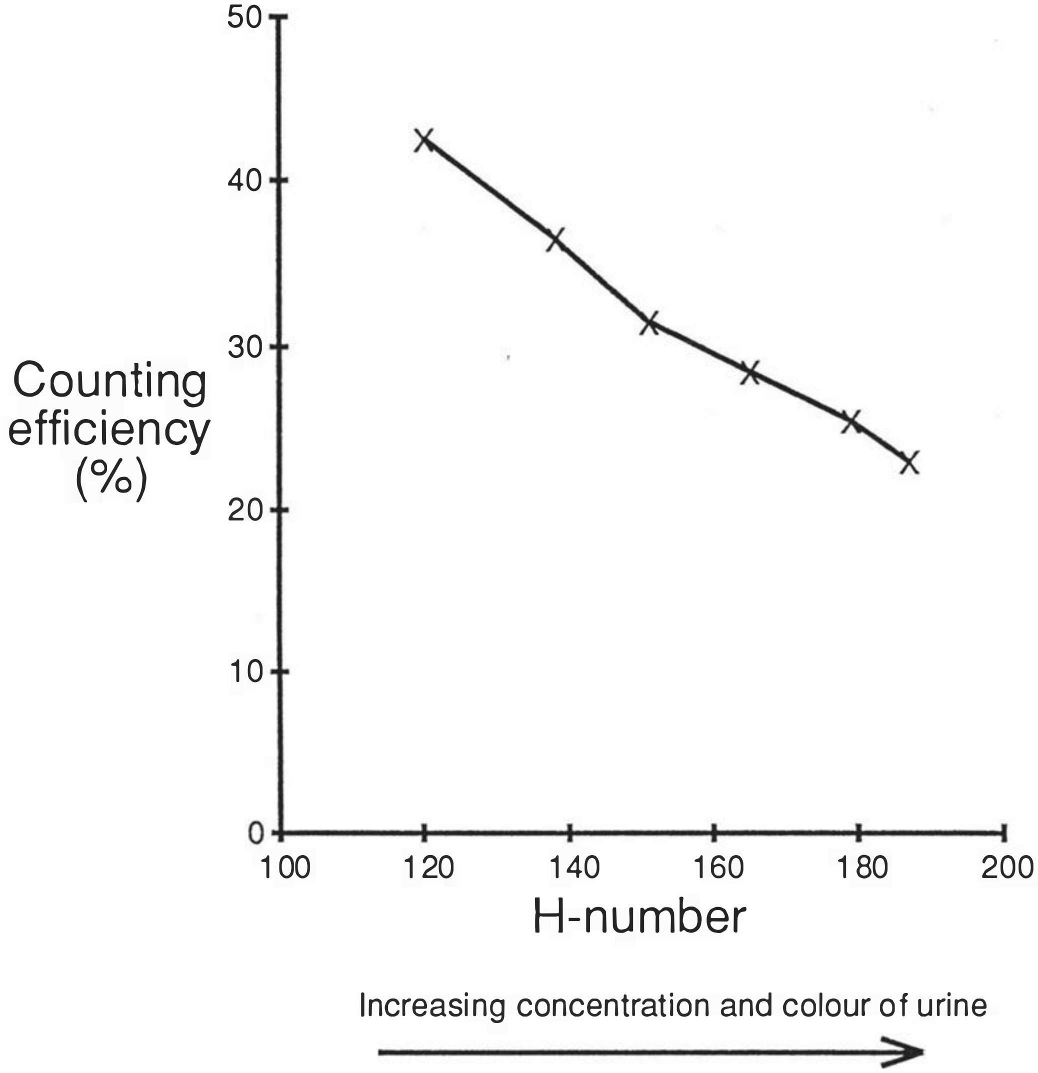


Figure 6.1 The relationship between H-number and counting efficiency of tritium in samples of leachate.

6.3 RESULTS

6.3.1 Measurement Of The Time Taken To Collect Leachate From The Urine Application

The lengths of time taken to collect each 25 g aliquot of leachate after the urine application are shown in Figure 6.2. The drainage patterns of the individual cores were markedly different even within the same soil type. For example, of the four Tokomaru cores (Figure 6.2a), core 6 drained very quickly; the first 25 g of leachate was collected within 24 seconds and 90% of the total leachate was collected within 3.5 minutes. The other cores were considerably slower, taking 5 hours to collect the first sample for core 14, and 22 hours to collect 90% of the total leachate. Presumably this range in flow rates resulted from variations in the physical characteristics of the soil cores. Soil structure, relative pore size and clay orientation affect water flow through soil (Thomas and Phillips, 1979), as do differences in surface microrelief which channel the water off ridges into cracks and small depressions (Cameron et al., 1979). In New Zealand soils, Mosley (1982) has shown that the distribution of macropores in the soil, such as those created by root networks, are highly variable and influence the rate of subsurface water flow.

An analysis of the variance of the times taken to collect the first 100 g of leachate indicated that there was no statistical difference between the mean collection times for each soil type (Table 6.2). There were, however, large differences between the times taken to collect 100 g of leachate from the fast and slow draining cores within each soil type (Figure 6.2). For statistical comparisons between soil types a better approach may have been to compare all the fast draining cores or all the slow draining cores. However, the number of fast and slow draining cores within each soil type varied from one to three which does not allow analysis of the sample variances. Instead regression analysis was used, where appropriate, to examine the proportion of the variation in a particular parameter (e.g., leachate composition) that was explained by the speed of drainage. For example, Figure 6.3 shows that the speed of drainage accounted for 87% of the variation in the relative activity of tritium in the first 25 g of leachate collected after the application of tritiated urine.

It was possible to use the data in Figure 6.3 to divide the cores into those which showed fast draining characteristics and those which showed slow draining characteristics. Between soil comparisons can only be made for the fast draining cores or the slow draining cores.

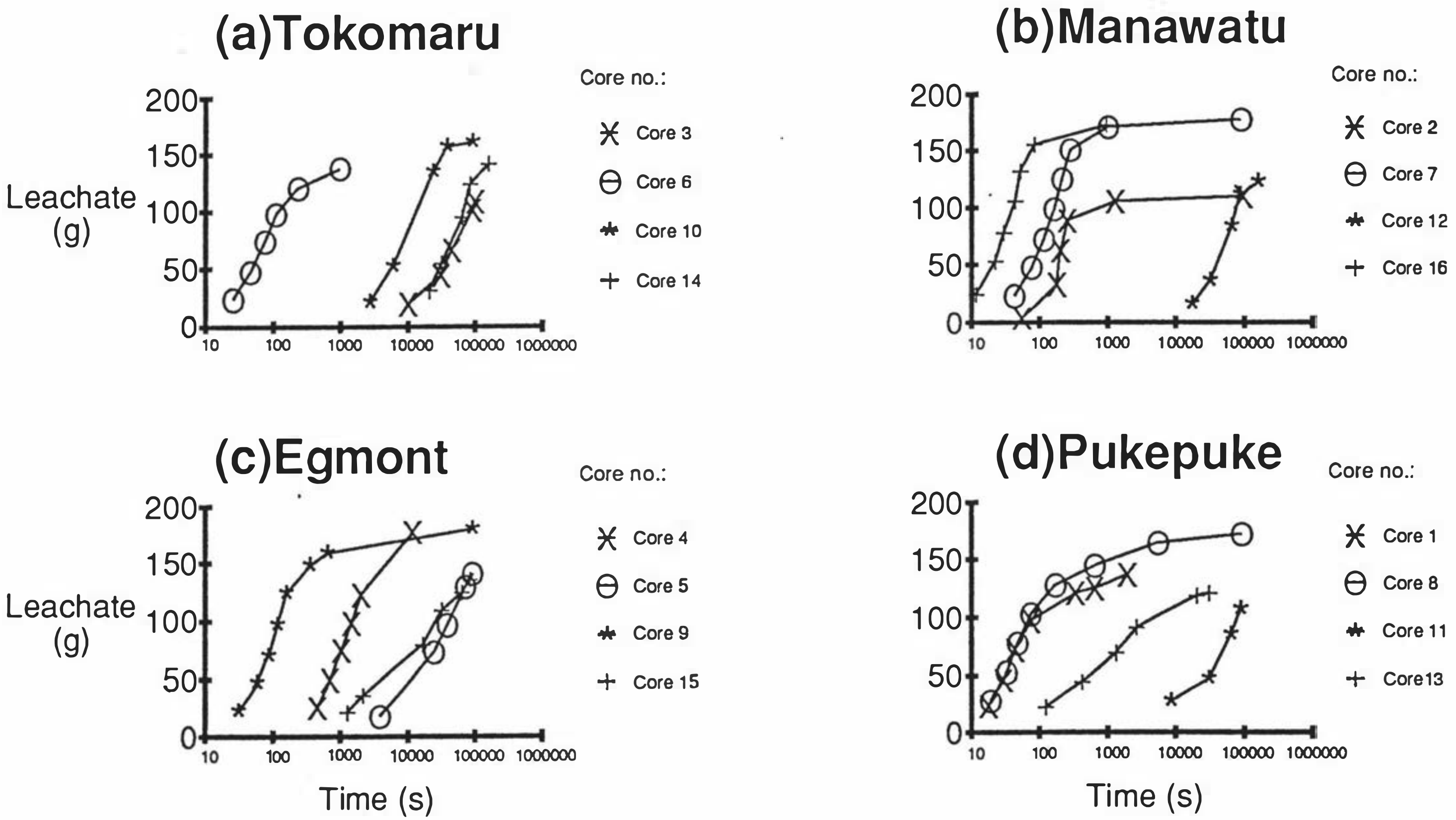


Figure 6.2 The length of time taken to collect 25 g aliquots of leachate following on application of dairy cow urine to intact soil cores.

Table 6.2 The analysis of variance table for the time (seconds) taken to collect 100 g of leachate (5.6 mm of drainage) from cores of four different soil types.

Source of variation	df	Sum of squares	Mean sum of squares	F
Blocks (replicates)	3	23.4×10^8	7.78×10^8	0.626 NS
Soil type	3	11.8×10^8	3.93×10^8	0.316 NS
Error	9	111.9×10^8	12.43×10^8	
Total	15	147.0×10^8		

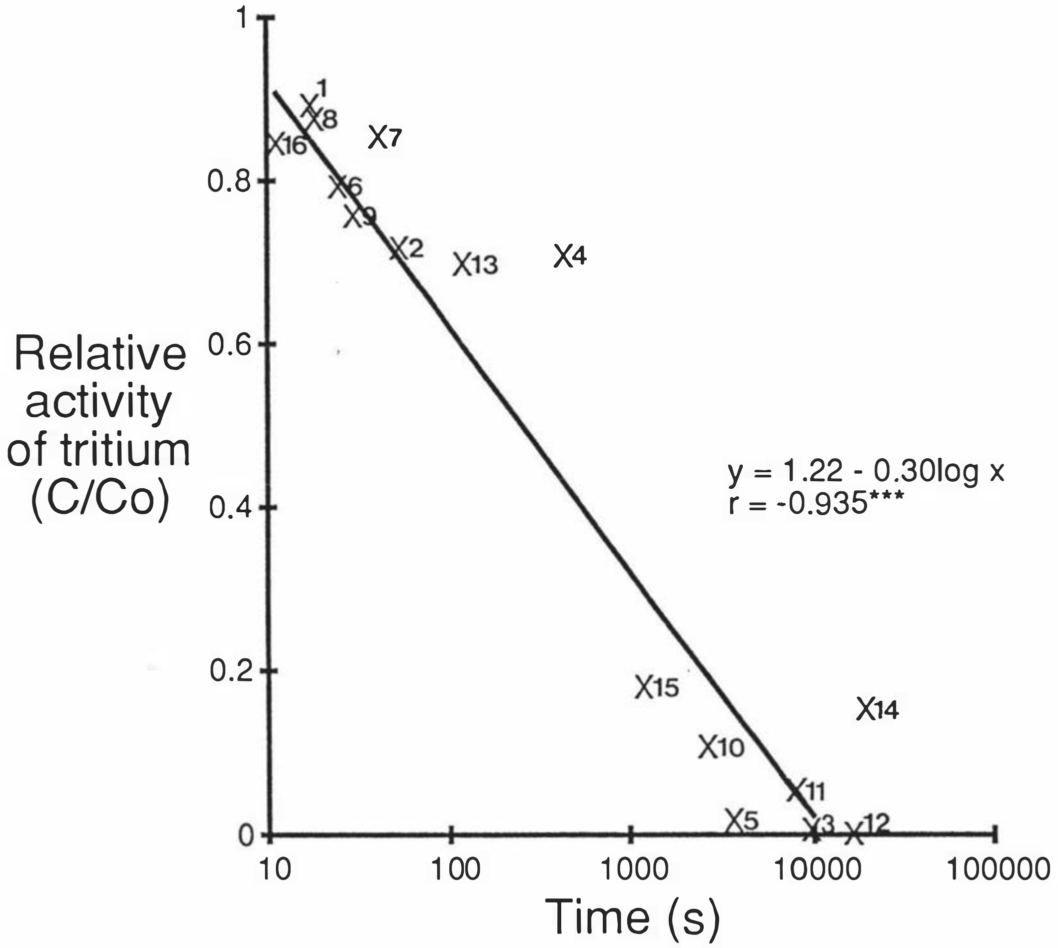


Figure 6.3 The relationship between the relative activity (C/Co) of tritium in the first 25 g of leachate collected after an application of tritium labelled dairy cow urine to intact soil cores and the time taken to collect that leachate (numbers on diagram represent core numbers).

The fast draining cores in Figure 6.3 were those for which the first 25 g of leachate were collected within 100 s from the time that the urine was applied, i.e., cores 1, 2, 6, 7, 8, 9 and 16 (Figure 6.3). The slow draining cores were those that required longer than 1,000 s to collect the first 25 g of leachate, i.e., cores 3, 5, 10, 11, 12, 14 and 15 (Figure 6.3). There were two exceptions which need elaboration. Core 13 has been included with the slow draining set of cores because although it was initially rapid draining, after the final rain event was applied the first 25 g of leachate were collected in 4600 s. This indicates that the rate of drainage from this core became slower with time, which was a characteristic of the other slow draining cores. The first 25 g of leachate from core 4 was collected in 540 s at the beginning and end of the experiment. The other cores which recorded similar times to collect 25 g of leachate at the start and end of the experiment were all fast draining cores, and so core 4 has been included with the fast draining cores.

To summarise, the cores were divided into two populations, fast (1, 2, 4, 6, 7, 8, 9 and 16) and slow draining cores (3, 5, 10, 11, 12, 13, 14 and 15). In the following discussion, where the results for a measured parameter were similar between all the cores within each population, the results from only one fast draining and one slow draining core are presented for each soil type.

For a statistical analysis of variance to be carried out to test differences between soil types for a particular parameter more replicates would have been required. The results from the experiment in Chapter 5, which used six replicates, indicated that four replicates would be sufficient (e.g., the coefficient of variation for the data on the amount of K leached from the simulated rain events was 13% with six replicates). However, there was a greater variability in the parameters measured in this experiment between the individual cores of the same soil type. As outlined above, this variability may be attributed to variations in the physical characteristics of soil cores, such as macropore distribution. The cores for this experiment were collected in early spring, whereas the cores for the previous experiment were collected in the summer. During the winter period, when the soil has a high moisture content for several months, the network of soil macropores may be reduced due to surface sealing and transport of sediment into the macropores (Beven and Germann, 1982), therefore resulting in a more variable pattern of water flow between replicate cores collected in the spring.

6.3.2 Chemical Analysis of Leachates from Urine Application and Rainfall Events

As discussed previously (section 6.3.1), there was a large variation in the time taken to collect the first 25 g of leachate from the urine application between the cores of each soil type. This variation was associated with differences in the activity of tritium and the concentration and quantity of K^+, Cl^-, NO_3^--N, NH_4^+-N and kjeldahl-N in the leachates from cores of the same soil type.

6.3.2.1 *The radioactivity of tritium in the leachates*

The ratio of tritium activity in the leachate (C) to the tritium activity applied in the urine (Co) is shown in Figure 6.4 for the four soils. This figure represents the data collected up to but excluding the final rain event, in which the rain was labelled with tritiated water.

For both fast and slow draining cores it was obvious that the tritiated urine flowed preferentially through the soil as evidenced by the amount of tritium activity recorded in the first leachate collected (Figure 6.4) when only 0.11-0.15 pore volumes had been applied (see section 2.7.1.2). There were differences in the extent to which preferential flow occurred between the fast and slow draining cores. For the fast draining cores the tritium activities in the first leachates collected were similar to that applied in the urine (Figure 6.4). In contrast, the initial tritium activities in the leachates from the slow draining cores were lower by at least a factor of ten (Figure 6.4). These differences between fast and slow draining cores imply that although some preferential flow occurred in all the soil cores, during slow drainage conditions the tritiated urine interacted with the initial soil water to a greater extent and therefore the tritium activity in the leachate was diluted.

If the breakthrough pattern showed no preferential flow (e.g., the theoretical pattern shown in Figure 6.4a it would have been possible to calculate the volume of soil water with which the tritium had exchanged. However, since the preferential flow of tritium through the soil involved uneven mixing with soil water, the result of this calculation would be difficult, if not impossible to interpret.

The relative activity of tritium in the leachate from the simulated rainfall events showed a gradual decline (Figure 6.4). During the collection of the leachate volume equivalent to one pore volume, the C/Co values decreased more rapidly for the fast draining cores compared with the slower draining cores (Figure 6.4). This indicated that less tritium was leached from

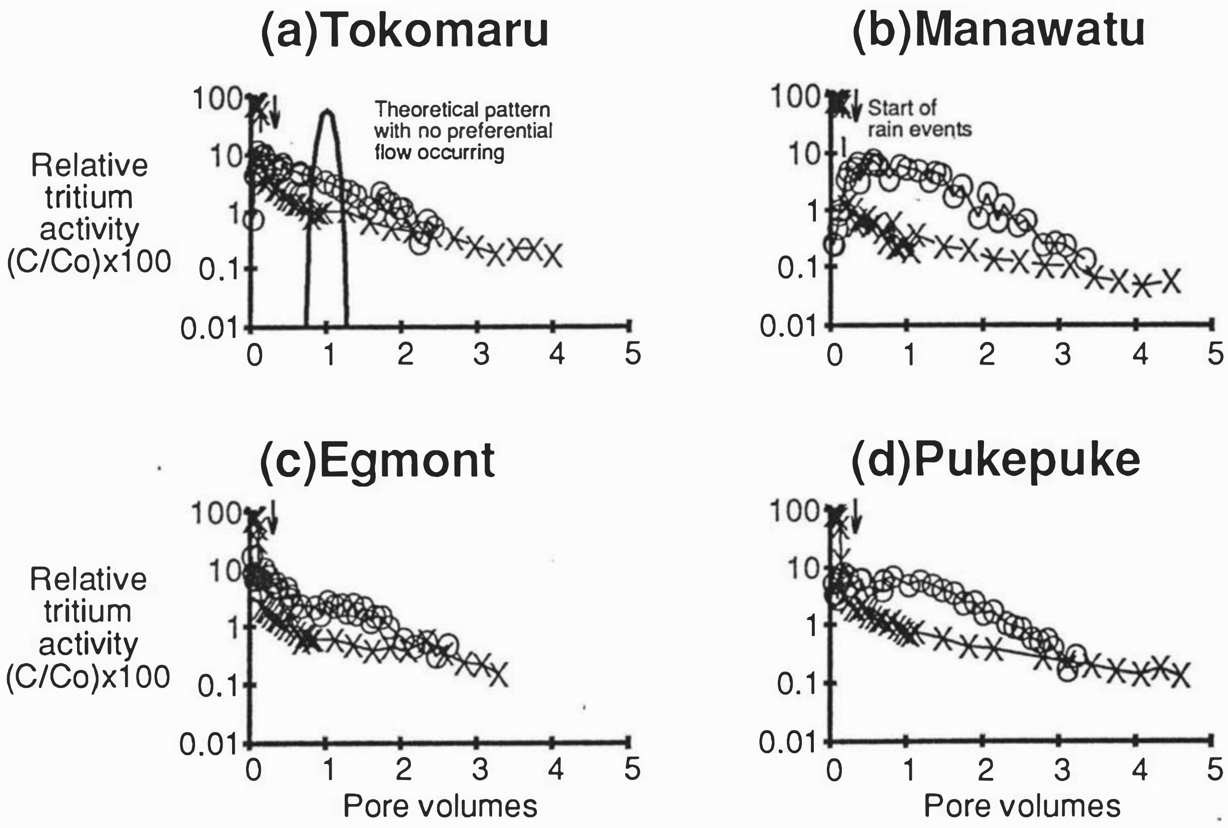


Figure 6.4 Breakthrough curves of tritium for fast draining (X) and slow draining (O) cores after the application of 200 ml of dairy cow urine and sufficient simulated rainfall to produce approximately 300 ml of drainage (equivalent to 2.4-4.6 pore volumes).

the rain events in the fast draining cores than from the slow draining cores. Thus in the fast draining cores 41-72% of the applied tritium was leached following urine application, but only 8-21% was leached subsequently from the rain events (Table 6.3). For the slow draining cores the reverse occurred with 0.3-33% of the applied tritium leached from the urine application and 47-63% leached from the rain events (Table 6.3). These results show that movement of urine into soil micropores which are not being affected by drainage flows may be a significant mechanism for retaining solutes such as K^+ in the soil by physical means.

6.3.2.2 *Potassium concentration in the leachates*

The changes in the relative K^+ concentration (C/Co) as the volume of leachate increased (Figure 6.5), showed a similar trend to the tritium activity (Figure 6.4). The K^+ concentration was initially at least ten times higher in the drainage from the fast draining cores than the slow draining cores, and was similar to the K^+ concentration in the applied urine. However, when the rain events began, this concentration declined rapidly and was in most cases ten times lower in the fast draining cores than the slower draining cores.

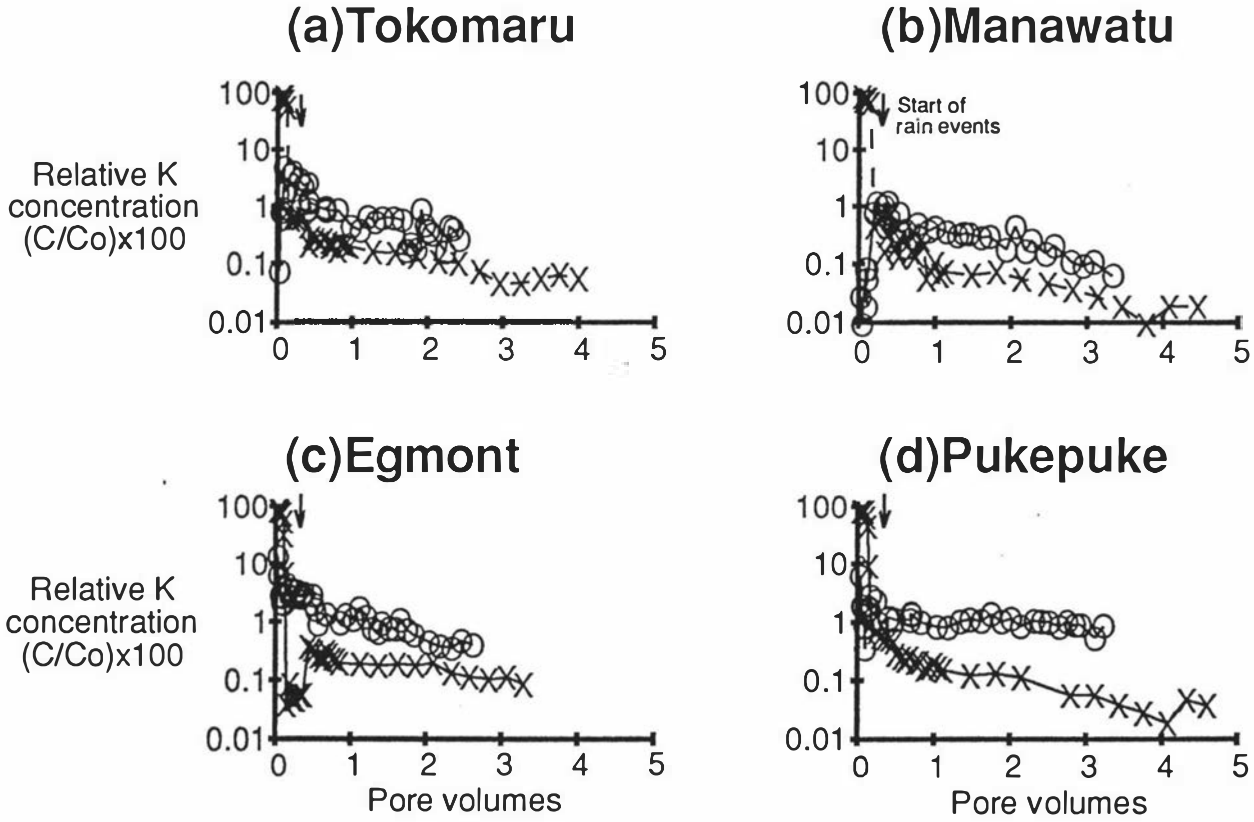
The amount of K leached immediately following the urine application in the fast draining cores ranged from 526 to 977 mg representing 40 to 74% of the K applied (Table 6.4). Compared with the fast draining cores a smaller quantity of K was leached from the slow draining cores (0.5-441 mg, 0.03-33% of that applied; Table 6.4). However, the slow draining cores tended to leach more K (7-27% of the K applied; Table 6.4) during the rain events compared with the fast draining cores (3-6% of the K applied, except for core 4 which leached 15%; Table 6.4). Despite this the total amount of K leached from both the urine application and the rain events was higher from the fast draining cores (46-78% of the K applied; Table 4.6) than the slow draining ones (7-30% of the K applied, except for core 4 which leached 45%; Table 6.4). Such a result supports the conclusions made in Chapter 5 that significant losses of urine K can occur from the top 15 cm of soil due to preferential flow of urine and that the subsequent leaching losses of K from rain events are comparatively lower.

6.3.2.3 *Chloride, nitrogen and calcium concentration of the leachates*

Initially higher concentrations of Cl^- were measured in the leachate from the fast draining cores compared with the slow draining cores. However, when the rain events began lower leachate Cl^- concentrations were measured from the fast draining cores (Appendix IVa, b). This trend was expected after observing the changes that occurred in the K^+ concentration of the leachates.

Table 6.3 The percentage of applied tritium that was leached from soil cores immediately following the urine application and during the simulated rain events.

Soil name Core no. (Drainage rate)	% of applied tritium in drainage After urine application	 After rain events	 Total	Mean
Tokomaru				
6 (fast)	54	21	75	
3 (slow)	4	52	56	64.0
10 (slow)	4	60	64	SEM = 4.0
14 (slow)	9	52	61	
Manawatu				
2 (fast)	41	12	53	
7 (fast)	71	10	81	67.5
16 (fast)	72	8	80	SEM = 7.5
12 (slow)	0.3	56	56	
Egmont				
4 (fast)	44	26	70	
9 (fast)	62	18	80	64.8
5 (slow)	4	52	56	SEM = 6.3
15 (slow)	6	47	53	
Pukepuke				
1 (fast)	53	14	67	
8 (fast)	71	16	87	69.5
11 (slow)	2	63	65	SEM = 6.1
13 (slow)	33	26	59	


(a)Tokomaru

(b)Manawatu

(c)Egmont

(d)Pukepuke

Figure 6.5 Breakthrough curves of K for fast draining (X) and slow draining (O) cores after the application of 200 ml of dairy cow urine and sufficient simulated rainfall to produce approximately 300 ml of drainage (equivalent to 2.4-4.6 pore volumes).

Table 6.4 The amounts of K leached from soil cores immediately following the urine application and during the simulated rain events (expressed as mg K core^{-1} and % of urine K applied).

Soil name	mg K in drainage			% of applied K leached			
Core no. (Drainage rate)	Urine application	Rain events	Total	Urine application	Rain events	Total	Mean
Tokomaru							
6 (fast)	749.9	59.2	809.1	57	4	61	
3 (slow)	16.7	199.4	216.1	1	15	16	26.8
10 (slow)	12.3	133.8	146.1	1	10	11	SEM=11.5
14 (slow)	118.5	134.6	253.1	9	10	19	
Manawatu							
2 (fast)	540.2	67.1	607.3	41	5	46	
7 (fast)	977.0	59.2	1036.2	74	4	78	50.8
16 (fast)	917.4	41.8	959.2	69	3	72	SEM=16.2
12 (slow)	0.5	94.7	95.2	0.03	7	7	
Egmont							
4 (fast)	525.8	199.4	725.2	40	15	55	
9 (fast)	862.7	84.0	946.7	65	6	71	43.5
5 (slow)	21.5	211.2	232.7	2	16	18	SEM=12.0
15 (slow)	43.9	361.8	405.7	3	27	30	
Pukepuke							
1 (fast)	750.3	84.6	834.9	57	6	63	
8 (fast)	797.9	50.1	848.0	60	4	64	49.0
11 (slow)	17.9	302.8	320.7	1	23	24	SEM=9.4
13 (slow)	441.1	153.4	594.5	33	12	45	

During the rapid preferential flow of urine through the soil core it was probable that the majority of the N in the drainage would still be in the form of urea rather than as products of hydrolysis (e.g., NH_4^+ and NO_3^-). Kjeldahl digestion of the leachates from the urine application confirmed that the majority of the N was present in an organic form, probably urea, and that very little had been hydrolysed to NH_4^+-N or NO_3^--N (Table 6.5).

Despite the fact that only a small proportion of urine N was converted to NH_4^+-N in the leachate that was collected immediately following the urine application (Table 6.5), the differences in NH_4^+-N concentrations between soil cores showed similar patterns to that of tritium, K^+ and Cl^-, with higher initial concentrations in the leachate from the fast draining cores (Appendix V). The leachates from the simulated rain events show the opposite pattern with higher NH_4^+-N concentrations in the leachates from the slow draining cores.

The accumulated amount of NO_3^--N in each leachate is presented in Figure 6.6. The concentration of NO_3^--N in the urine at the time of application was negligible (Table 6.1), and the amounts native to the soil were small (Figure 5.7), thus the amounts of NO_3^--N in the drainage were also small initially (Figure 6.6). Following urine application, the quantity of NO_3^--N in the leachates increased with time as the urea-N was hydrolysed to NH_4^+-N and was then nitrified to NO_3^--N (Helyar, 1976). The build up in the amount of NO_3^--N in the leachate was rapid in the fast draining cores, with a noticeable increase after 40 mm (0.41-0.55 of a pore volume) of drainage which was collected between 4 and 7 days after urine application (Figure 6.6). In the slow draining cores the build up of NO_3^--N in the leachates was much slower (Figure 6.6).

Considerable amounts of Ca (43-371 mg Ca core^{-1}) were leached from the soil cores, representing 2-13% of the initial soil exchangeable Ca^{++} content (Table 6.6), presumably following displacement from the soil surface exchange sites by K^+ and NH_4^+ from the urine. These amounts of Ca are higher than those measured in the previous experiment (where the amount of soil exchangeable Ca^{++} leached was 0.2-1%; section 5.3.8) probably because smaller amounts of simulated rainfall (90 mm) were applied during that experiment. Differences in the amounts of Ca leached from the fast and the slow draining cores immediately following the urine application were not as obvious as for the different amounts of K leached or the other elements already discussed. For the fast draining cores, similar amounts of Ca (19-52 mg Ca core^{-1}; Table 6.6) were leached from each soil type during the simulated rain events. However, greater amounts of Ca were leached from the Tokomaru and Manawatu slow draining cores (140-368 mg Ca core^{-1}; Table 6.6) than leached from the

Table 6.5 The amounts and forms of N leached from soil cores immediately following the urine application (mg N core^{-1}).

Soil name Core no. (Drainage rate)	NH_4^+-N	NO_3^--N	Kjeldahl-N	Total N leached as a % of applied N
Tokomaru				
6 (fast)	68	<0.01	530	48
3 (slow)	7	0.04	66	6
10 (slow)	4	0.08	28	3
14 (slow)	19	0.08	83	8
Manawatu				
2 (fast)	181	<0.01	37	34
7 (fast)	129	<0.01	685	62
16 (fast)	459	<0.01	489	45
12 (slow)	1	0.30	4	0.4
Egmont				
4 (fast)	54	<0.01	407	37
9 (fast)	229	0.05	607	55
5 (slow)	9	0.08	33	3
15 (slow)	32	0.73	43	4
Pukepuke				
1 (fast)	134	<0.01	502	46
8 (fast)	147	0.02	639	58
11 (slow)	5	0.07	18	2
13 (slow)	12	<0.01	333	30

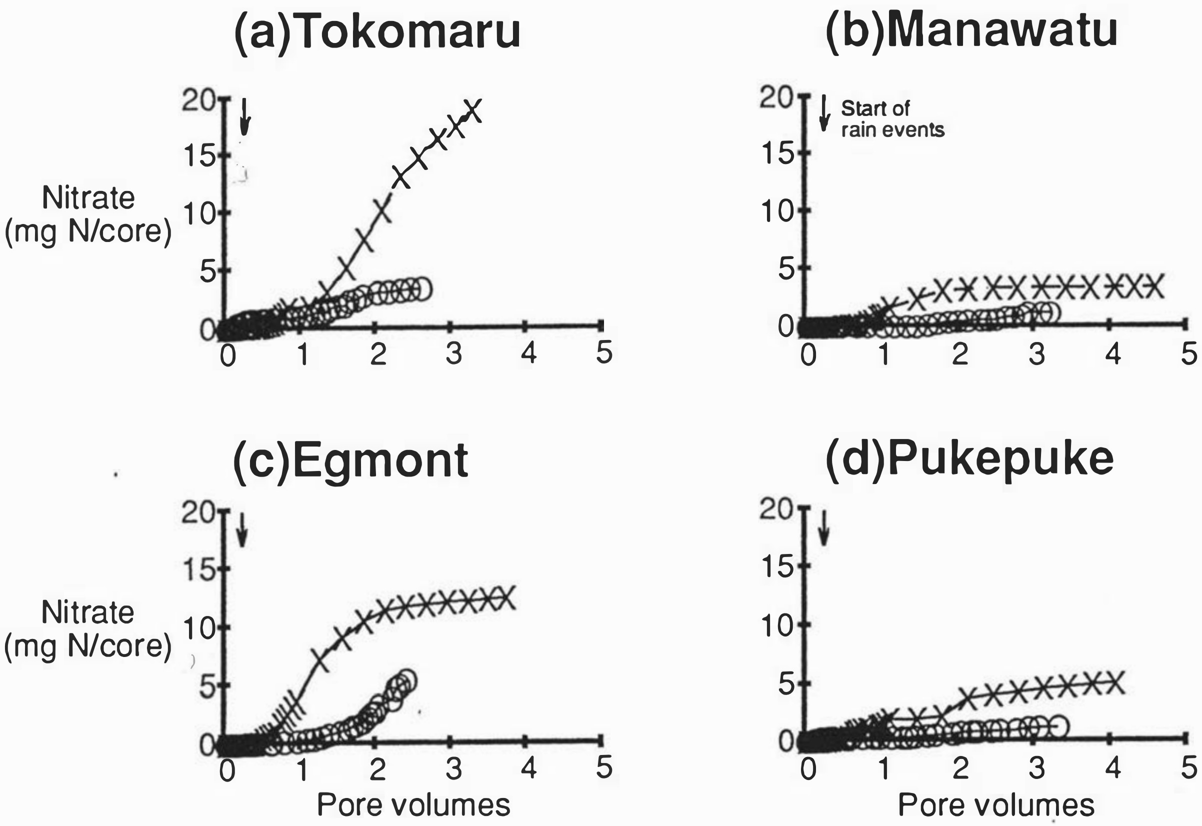


Figure 6.6 The accumulated amount of NO_3^--N leached from fast draining (X) and slow draining (O) cores after the application of 200 ml of dairy cow urine and sufficient simulated rainfall to produce approximately 300 ml of drainage (equivalent to 2.4-4.6 pore volumes).

Table 6.6 The amounts of Ca leached from soil cores immediately following the urine application and the simulated rain events (expressed as mg Ca per $core^{-1}$ and as a % of the amount of exchangeable Ca initially present in the soil).

Soil name Core no. (Drainage rate)	mg Ca leached per core Urine application	 Rain events	Total	% of initial soil Ca
Tokomaru				
6 (fast)	14	44	58	2.3
13 (slow)	26	197	223	8.5
10 (slow)	4	140	144	5.3
14 (slow)	10	298	308	10.4
Manawatu				
2 (fast)	27	31	58	1.8
7 (fast)	34	52	86	2.9
16 (fast)	34	19	53	1.6
12 (slow)	3	368	371	12.8
Egmont				
4 (fast)	31	45	76	3.8
9 (fast)	8	37	45	1.7
5 (slow)	14	73	87	4.0
15 (slow)	6	33	39	1.6
Pukepuke				
1 (fast)	51	17	68	2.6
8 (fast)	41	33	74	2.4
11 (slow)	1	145	146	5.5
13 (slow)	5	38	43	1.8

Egmont and Pukepuke slow draining cores (33-145 mg Ca core^{-1}; Table 6.6). This reflects the greater amounts of native exchangeable Ca^{++} in the Tokomaru and Manawatu soil types (Table 5.5).

6.3.3 Leachate from the Final Rain Event

Prior to the final rain event the tritium activity in the leachates from most soil cores had fallen to <1000 dpm ml^{-1}. The activity of the tritium added to the final rain event was therefore adjusted to be 600 times greater than this activity. The relative activity (C/Co) of the tritium in the first 25 g of leachate collected from the final rain event is shown in Table 6.7. The appearance of high activities of tritium in the leachates indicates that there was some preferential flow of the rain water in all of the fast draining cores and in some of the slow draining cores (cores 10, 11 and 13) as well. These results confirm that macropore flow was consistently higher in the fast draining cores and that, with the exception of cores 10, 11 and 13, the flow characteristics of a core were slow to change as they were consistent throughout the experiment.

As discussed previously (section 6.3.1), rapid preferential flow occurred at the start of the experiment in core 13 and this was still evident in the final rain event although the rate of drainage had become slower during the experiment. For cores 10 and 11 the relative activities of tritium in the leachate from the final rain event (0.59 and 0.31 respectively; Table 6.7) were higher than the initial relative activities of tritium in the leachate from the urine application (0.11 and 0.06 respectively; Table 6.7). This suggests that preferential flow at the end of the experiment was greater than at the start. In the case of core 10 this may have been due to the development of a macropore network for preferential flow of the rain water during the experiment through the activity of earthworms. Casting activity of earthworms was observed on the surface of the Tokomaru, Manawatu and Egmont soil types. For core 11 there was a gradual change in water content from 0.45 g g^{-1} at the start to 0.64 g g^{-1} at the end of the experiment suggesting that preferential flow may have been greater at the final rain event due to the higher moisture content of the core. Increased preferential flow rates with increased soil moisture contents have been measured before by Hoogmoed and Bouma (1980) and Scotter and Kanchanasut (1981). There may have been some piston displacement of core water occurring in the slow draining cores 3, 5, 12, 14 and 15, as the tritium activity in the leachate from the final rain event was very low (Table 6.7) and may have originated from the initial urine activity remaining in the core rather than the freshly added tritium in the rain application.

Table 6.7 The relative activity (C/Co) of tritium in the first 25 g of leachate collected after the original urine application, and after the application of the final rain event.

Soil name Core no. (Drainage rate)	Original urine application	Final rainfall
Tokomaru		
6 (fast)	0.79	0.68
3 (slow)	0.01	0.01
10 (slow)	0.11	0.59
14 (slow)	0.16	0.01
Manawatu		
2 (fast)	0.72	0.84
7 (fast)	0.86	0.84
16 (fast)	0.85	0.63
12 (slow)	0.003	0.01
Egmont		
4 (fast)	0.71	0.20
9 (fast)	0.76	0.33
5 (slow)	0.02	0.06
15 (slow)	0.18	0.11
Pukepuke		
1 (fast)	0.90	0.34
8 (fast)	0.88	0.39
11 (slow)	0.06	0.31
13 (slow)	0.70	0.72

6.3.4 Soil Analysis

The following results were obtained from chemical analysis of the soil cores prior to and after the completion of the experiment. After urine deposition and the subsequent rain events, the amounts of soil exchangeable K^+, and to a lesser extent water soluble K^+, were higher than the amounts present in the original soils (Figure 6.7). The gains in soil exchangeable K^+ were most pronounced in the slow draining cores reflecting the smaller amounts of preferential flow which occurred immediately following urine application. The increases in exchangeable soil K^+ were detectable at all soil depths. The slower draining cores showed a proportionately greater gain in soil exchangeable K^+ at the lower soil depths 5-10 and 10-15 cm) compared with the fast draining cores (Figure 6.7). In both the slow and fast draining cores the greatest gain in soil exchangeable K^+ occurred in the topsoil (0-5 cm; Figure 6.7), a result that was similar to that obtained in the field experiment on the Tokomaru silt loam (Figure 3.3).

The distribution of exchangeable NH_4^+-N throughout the soil cores (Figure 6.8) followed a similar pattern to the exchangeable K^+ results (Figure 6.7). The greatest gains in exchangeable NH_4^+-N occurred in the 0-5 cm depth of soil, and at all soil depths the slow draining cores showed larger gains in exchangeable NH_4^+-N (Figure 6.8).

For most cores the soil pH at the end of the experiment was higher than at the beginning before urine application (Table 6.8). This increase in soil pH is likely to have been brought about by the enzymatic hydrolysis of the urea from the urine (urea-N accounts for 70% of the total urine-N; Doak, 1952). As explained in sections 4.1, 4.3.1 and 5.3.4, hydrolysis of urea to NH_4^+ and HCO_3^- initially increases soil pH, and then nitrification of NH_4^+ to NO_3^- and H^+ subsequently decreases soil pH (Doak, 1952; Helyar, 1976). The slow draining cores showed a larger increase in soil pH during the experiment compared with the fast draining cores (Table 6.8). This indicates that the urine N was converted to NH_4^+ but not to NO_3^- in the slow draining cores. Further evidence for this lack of nitrification is provided by the greater concentration of NH_4^+ in the leachate (Appendix V) and the smaller amount of NO_3^- leached (Figure 6.6) from the slow draining cores compared with the fast draining cores.

The inhibition of nitrification in the slow draining cores may have resulted initially from the rapid increase in soil pH immediately after urine application, an effect described by Doak (1952) and Broadbent et al. (1958). Larger amounts of urine N were retained in the slow draining cores (up to 1099 mg N per core; Table 6.5). The higher the initial urea-N content of the core, the higher the pH will rise after the urea is hydrolysed. High soil pH's (pH >8; Doak, 1952) can inhibit the *Nitrobacter* species that are important for nitrification (Stojanovic and

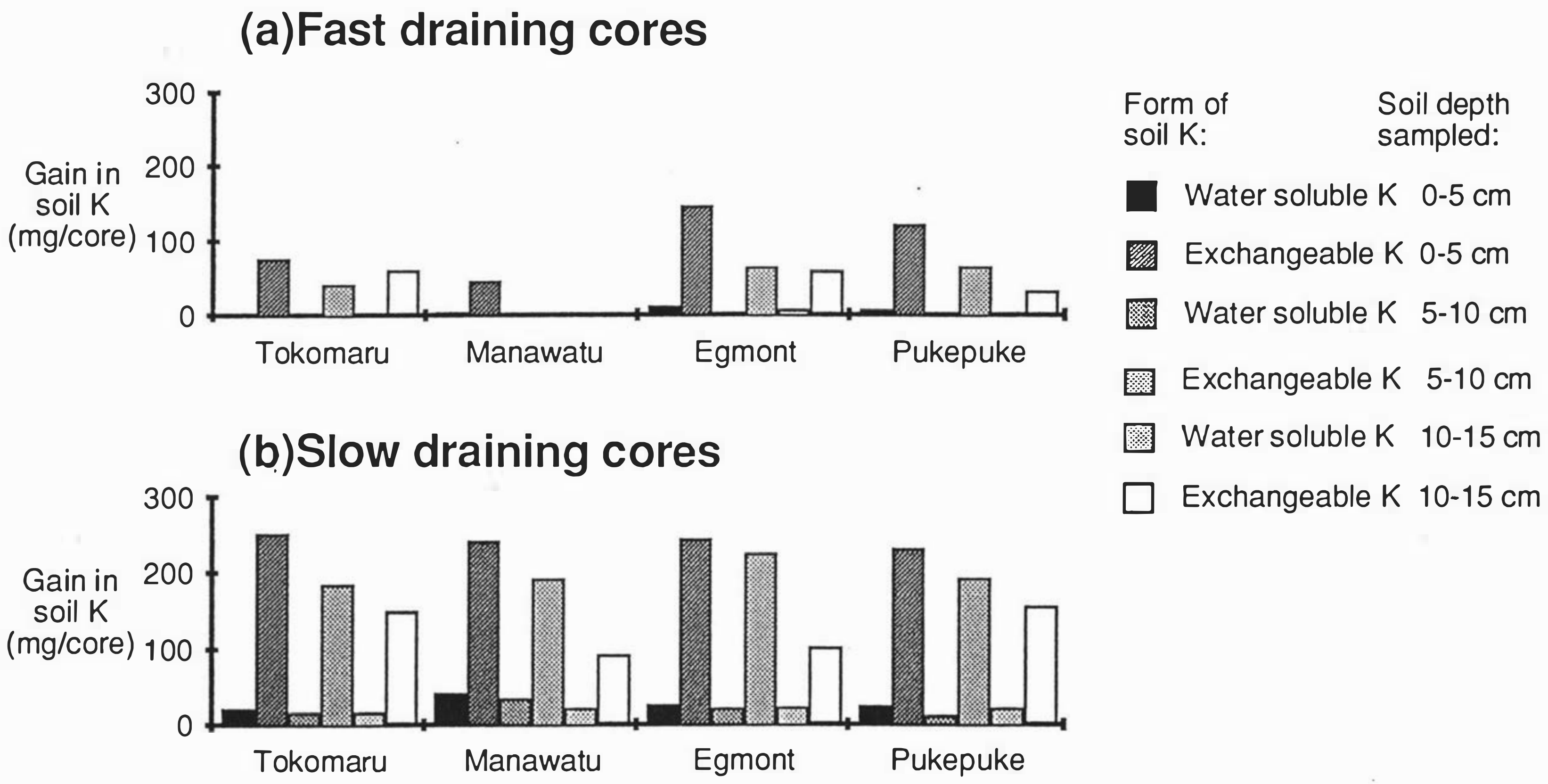


Figure 6.7 The gain in water soluble K^+ and exchangeable K^+ in soil cores after the application of urine and simulated rainfall (i.e., K content [amended soil - original soil]).

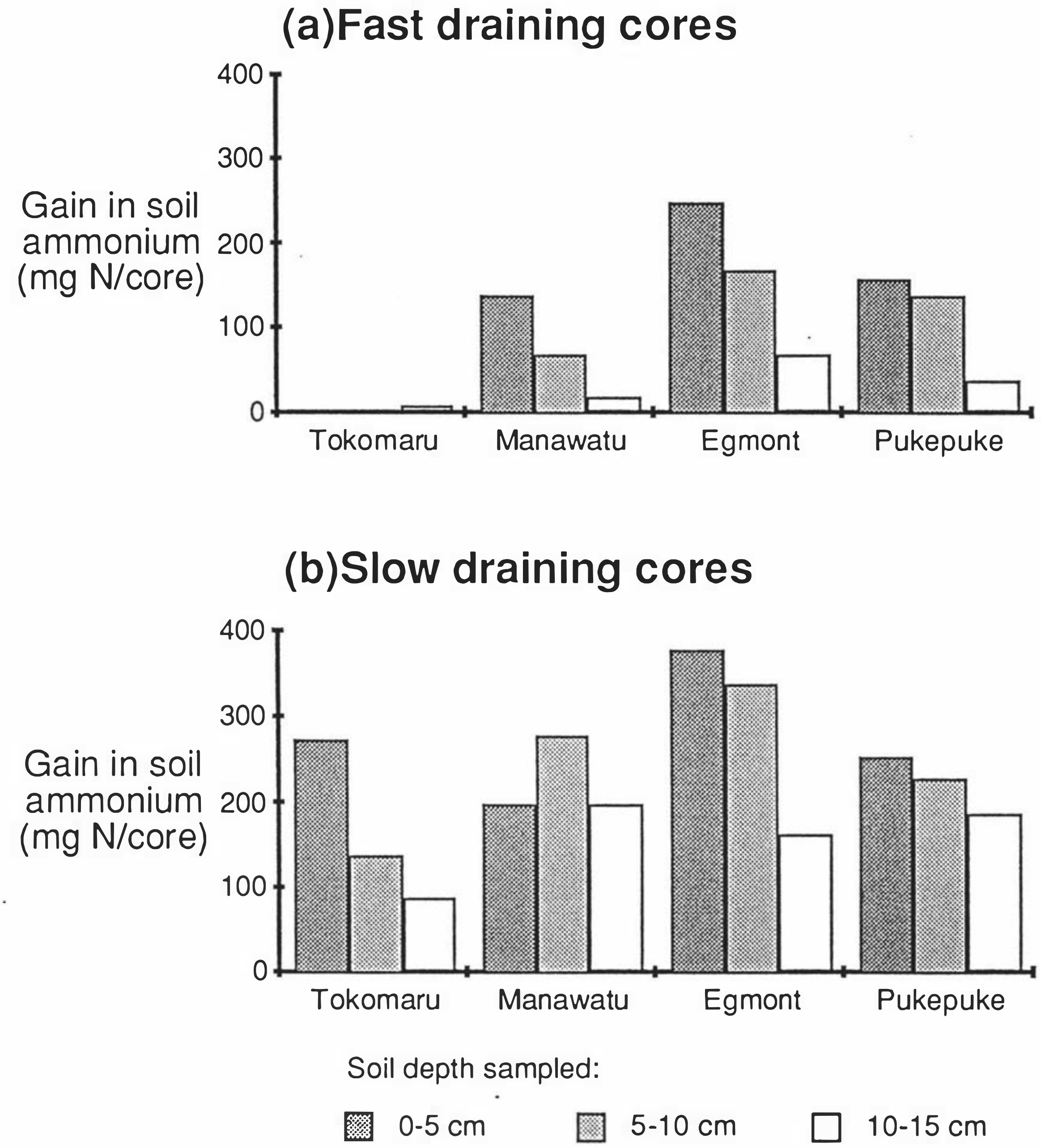


Figure 6.8 The gain in KCl-extractable soil NH_4^+-N in soil cores after the application of urine and simulated rainfall (i.e., NH_4^+-N content [amended soil - original soil]).

Table 6.8 Soil pH before and after the experiment (mean of all depths).

Soil name Core no. (Drainage rate)	Initial	Final
Tokomaru		
6 (fast)	5.9	5.8
3 (slow)	5.6	6.2
10 (slow)	5.8	6.0
14 (slow)	5.9	6.3
Manawatu		
2 (fast)	5.8	6.2
7 (fast)	5.9	6.1
16 (fast)	6.0	6.0
12 (slow)	6.0	6.6
Egmont		
4 (fast)	5.4	5.8
9 (fast)	5.6	6.0
5 (slow)	5.4	6.4
15 (slow)	5.6	6.3
Pukepuke		
1 (fast)	5.8	5.8
8 (fast)	5.7	5.8
11 (slow)	5.7	6.3
13 (slow)	5.8	6.2

Alexander, 1958). The high concentration of other ions such as Cl^- in the urine may also have decreased nitrification (Campino, 1982) in the slow draining cores, as these cores also retained more Cl from the urine application compared with the fast draining cores (Appendix IVb).

Another factor which may have reduced the nitrification rate of the slow draining cores was the supply of oxygen (Whitehead, 1970). At the end of the experiment only 0-10% of the porosity of the slow draining cores was filled with air, the remainder being filled with water. This would have limited the oxygen supply for nitrification severely.

The change in soil pH that occurred throughout the experiment is likely to be a function of the difference between the amount of NH_4^+-N and NO_3^--N produced, because unless NH_4^+ is nitrified or reutilised by plants no H^+ is generated to neutralise the HCO_3^- or OH^- produced during urea hydrolysis (Helyar, 1976). This relationship is shown in Figure 6.9, where it is assumed that the presence of 1 mmole of NH_4^+ is associated with the production of 1 mmole of OH^-, whereas the conversion of 1 mmole of NH_4^+ to NO_3^- is associated with the production of 2 mmoles of H^+ (Helyar, 1976). The difference between the NH_4^+-N and the NO_3^--N produced for each soil core is the sum of the amounts of NH_4^+-N and NO_3^--N that were leached throughout the experiment plus the increase in soil exchangeable NH_4^+-N between the beginning and end of the experiment. The relationship in Figure 6.9 is essentially a buffer curve of the amount of OH^- or H^+ added versus the change in soil pH. A logistic curve of the type $y=A/[1+B^*\exp(C^*x)]$ was fitted to the data, and the resultant equation accounted for 78% of the variation.

It can be seen from Figure 6.9 that the slow draining cores were the ones with the least amount of NO_3^--N produced and so had the greatest increase in soil pH. In the field situation, low rates of nitrification in the soil are likely to occur during conditions of poor drainage or a perched water table. Urine depositions on these areas are likely to result in higher soil pH, smaller K leaching losses from the urination event, and less leaching of NO_3^- (through the lack of nitrification and the retention of NH_4^+ by the soil surface exchange sites) than when urine is deposited on free draining sites.

6.3.5 **Potassium Balance**

The quantitative redistribution of the urine K into the amounts leached from the initial urine application, the amounts leached during the subsequent rainfall events, the amounts taken up by the herbage and the amounts remaining in the soil at the end of the experiment is

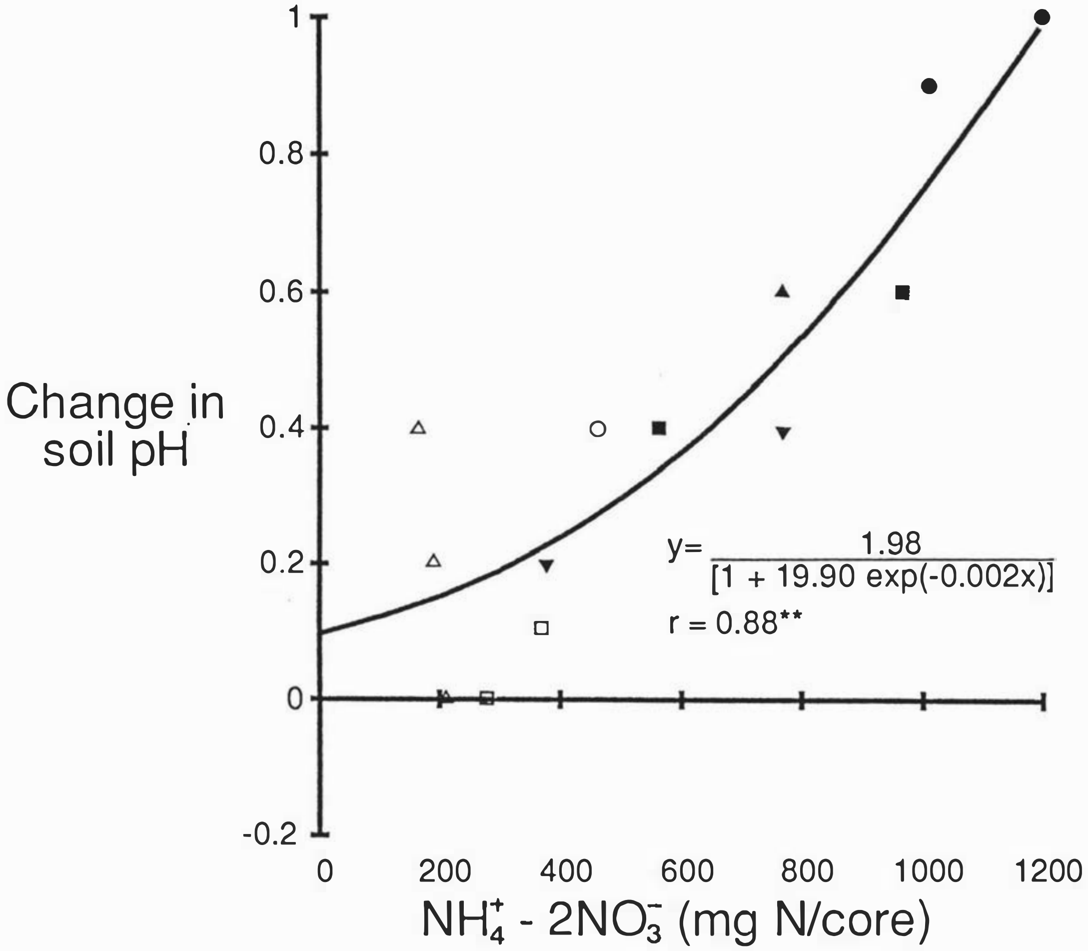


Figure 6.9 The relationship between change in soil pH and the difference between the total amounts of NH_4^+-N and NO_3^--N produced in and leached from fast (open symbols) and slow (closed symbols) draining intact cores of Tokomaru (▽), Manawatu (△), Egmont (○) and Pukepuke (□) soil.

shown in Figure 6.10. All the data is expressed as a proportion of the amount of urine K applied.

On average, the quantitative redistribution of urine K into these fractions was similar between all soil types, however there were differences between fast and slow draining cores. The fast draining cores tended to have a higher proportion of K loss occurring in the leachate from the urine application and a smaller proportion remaining as soil at the end of the experiment. The proportion of K leached from the rain events and taken up in the herbage was similar between all soils and together accounted for less than 30% of the urine K applied.

Overall between 67% and 110% of the applied urine K was recovered as either leachate K, soil K or plant K (Figure 6.10). These recoveries are comparable with the recoveries reported in Chapter 5. Reasons for the less than 100% recovery have been discussed previously (section 5.3.1).

6.4 DISCUSSION

6.4.1 Preferential Flow of Urine Following Application

The presence of high tritium activity and high concentrations of K^+, Cl^- and NH_4^+-N in the drainage collected immediately after the urine application (Figures 6.4, 6.5; Appendices IV, V) indicates that the urine was flowing preferentially through the soil cores. Such flow can occur through a network of large pores, channels, cracks, root holes, earthworm holes and ped faces that exist in the soil (Anderson and Bouma, 1973; Kissel et al., 1973; Bouma and Dekker, 1978; Scotter and Kanchanasut, 1981; Tyler and Thomas, 1981).

Preferential flow after application of urine through the soil resulted in losses of 72, 74, 80 and 62% of the applied tritium, K, Cl and N respectively from the 15 cm deep soil cores (Tables 6.3, 6.4, 6.5; Appendix IVb). These results are similar to those obtained in a field experiment by Quisenberry and Phillips (1976). They found that after applying 42 mm of Cl labelled water to a silt loam soil with a moisture content at field capacity, 80% of the Cl moved beyond a soil depth of 90 cm within 24 hours.

In the fast draining cores of this experiment there was initially very little dilution of the applied tritium, K and Cl in the leachate (Figures 6.4, 6.5; Appendix IVa). This lack of dilution can be attributed to the applied urine bypassing the soil water, hence very little of the soil water was displaced, and there was limited opportunity for the urine solutes to interact with the

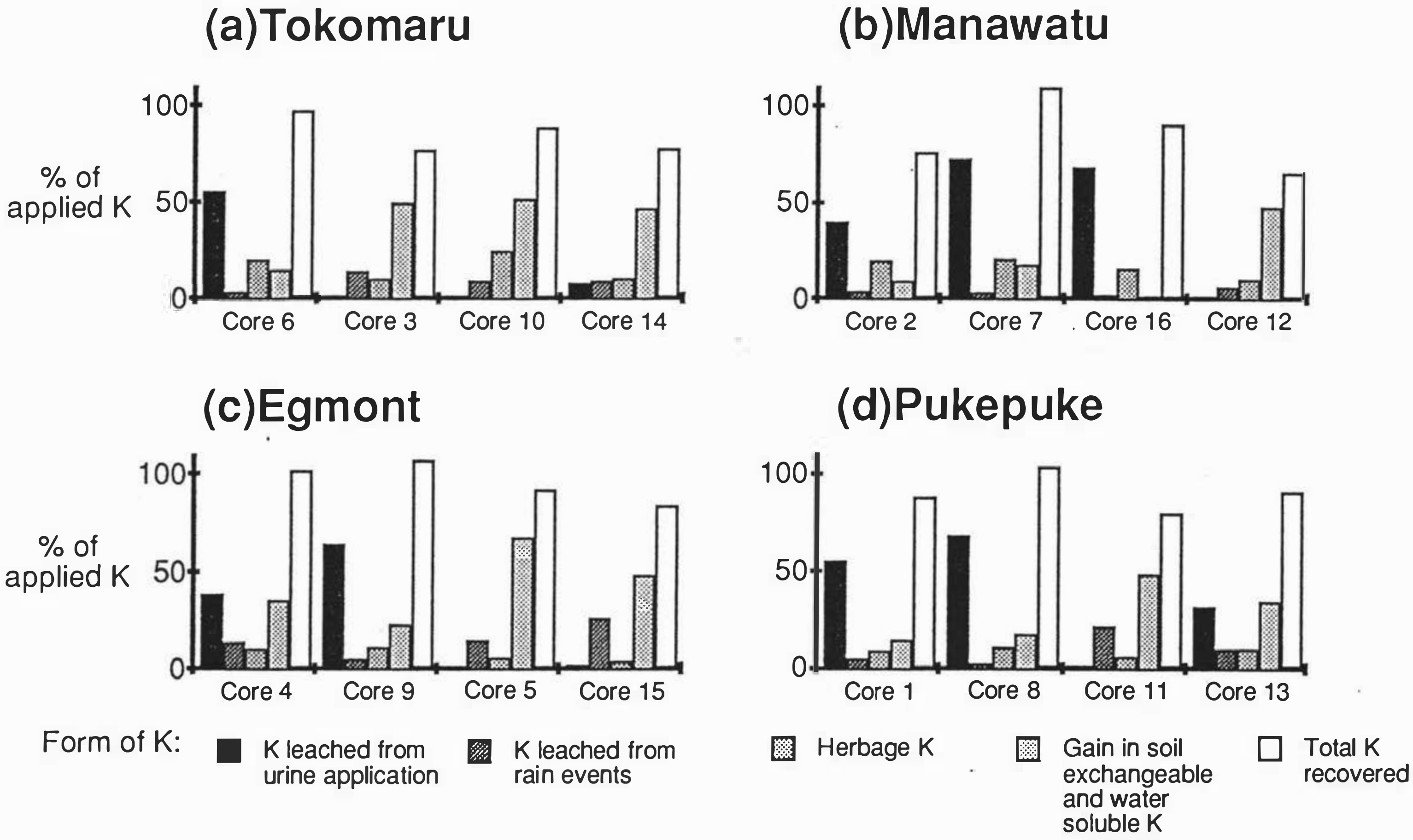


Figure 6.10 The amounts of leachate, soil and plant K present following an application of cow urine to intact cores of different soil types. The amounts of K are expressed as a % of the amount of urine K applied.

resident soil water. White et al. (1986) drew similar conclusions after measuring the movement of Cl through a structured clay soil. Lower concentrations of K^+, Cl^- and $NH_4{}^+$-N, and activities of tritium were measured in the leachate of the slow draining cores (Figures 6.4, 6.5; Appendices IV, V). This indicated that there was more flow of the urine through the soil matrix and hence more of the resident soil water was displaced from the slow draining cores.

The similarity in the C/Co curves for tritium and K (Figures 6.4, 6.5) suggested that there was a relationship between the proportions of the applied tritium and K that were collected in the leachate. Regression analysis demonstrated a highly significant 1:1 relationship between the proportions of applied tritium and K that were leached from each soil core (Figure 6.11a). When tritium and kjeldahl-N, and tritium and Cl were compared together similar relationships were also obtained (Figures 6.11b, 6.11c). Despite the differences in soil chemical properties between the four soil types (Table 5.1), these 1:1 relationships were highly significant for each soil and for each solute (Table 6.9).

Although Figure 6.11a shows that similar amounts of tritium and K were leached immediately following the urine application from each soil core, it is perhaps more relevant to compare the effect of the rate of drainage on the relative activity of tritium and the relative concentration of K^+ in the leachate. For the fast draining cores there appears to be little difference between the pattern of occurrence of tritium and K^+ in the leachate (Figure 6.12a). The higher values for the relative K^+ concentration compared with the relative activity of tritium in the initial leachate from the Egmont soil probably resulted from the presence of some native soil solution K^+ in the first leachate collected. In contrast, for the slow draining cores the relative activity of tritium is consistently higher than the relative K^+ concentration (Figure 6.12b). While it is impossible to determine if the K in the leachate in Figure 6.12b is from the urine application or is native soil solution K^+, these data suggest that at the slower drainage rate the K was being adsorbed by the soil while the tritiated water in the urine was moving through the soil core unhindered. This difference is not obvious in Figure 6.11a because the total amount of tritium and K leached from the slow draining cores was very low (<6% of the tritium applied and <3% of the total K applied).

The data presented in Figure 6.11 shows that when urine moves rapidly through the soil by preferential flow there is little opportunity for significant exchange to take place between the ions in the urine and the ions on soil surfaces regardless of whether the solute is strongly adsorbed (K^+), weakly adsorbed by the soil (urea; Broadbent et al., 1958) or not adsorbed by the soil (Cl^-). Similarly, under such conditions the usually strongly sorbed phosphate ion is not adsorbed by the soil and can move rapidly through the soil by preferential flow (Scotter, 1978; Scotter and Kanchanasut, 1981).

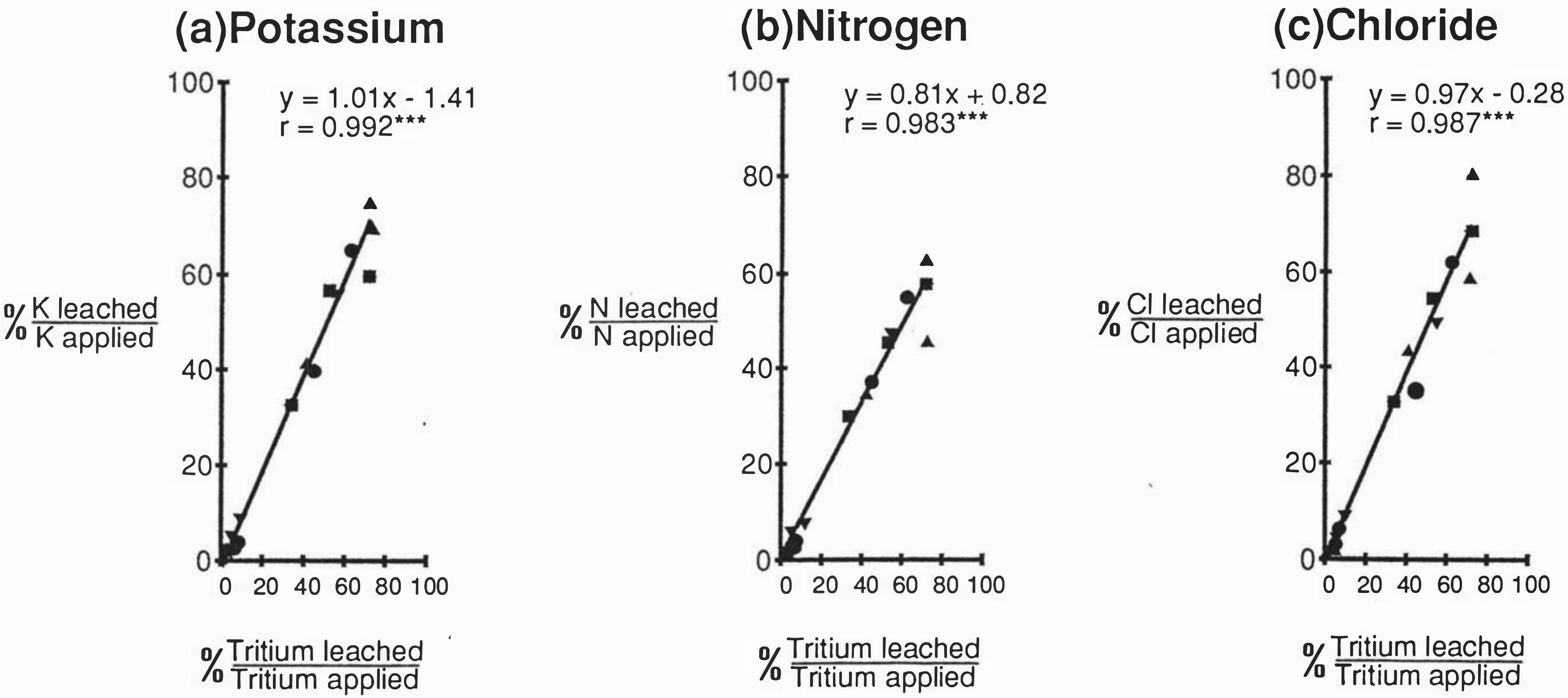
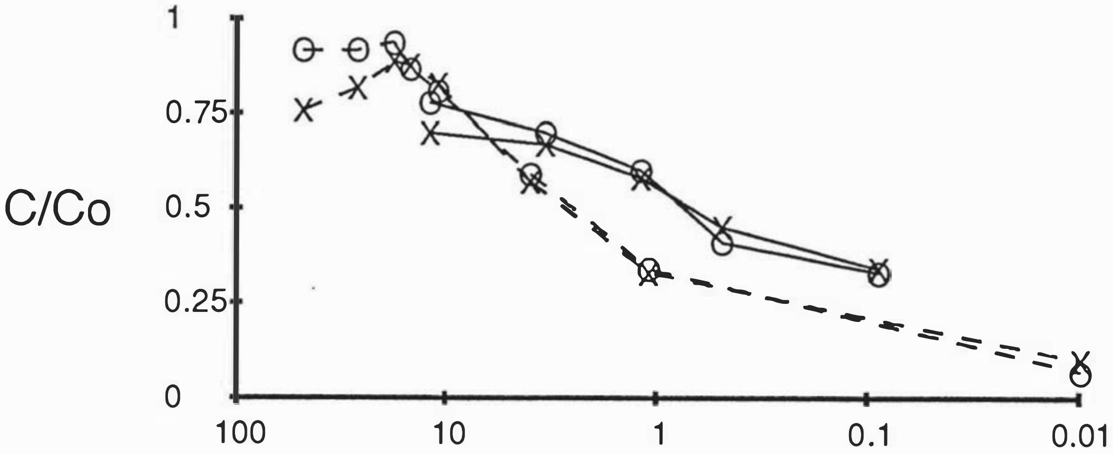


Figure 6.11 The relationship between the proportion of tritium leached and the proportion of (a) K, (b) total N and (c) Cl leached immediately after the application of tritium labelled dairy cow urine to intact cores of Tokomaru (▽), Manawatu (△), Egmont (○) and Pukepuke (□) soil.

Table 6.9 Regression equations for the relationship between the % tritium (^{3}H) leached and the % K, Cl and N leached immediately following the urine application.

Soil name	Regression equation	r
Tokomaru	% K = 1.11 x % ^{3}H - 2.67	0.999***
	% Cl = 0.94 x % ^{3}H - 0.71	0.999***
	% N = 0.88 x % ^{3}H + 0.39	0.998**
Manawatu	% K = 1.00 x % ^{3}H - 0.19	0.997**
	% Cl = 0.95 x % ^{3}H + 1.57	0.965*
	% N = 0.74 x % ^{3}H + 1.37	0.957*
Egmont	% K = 1.06 x % ^{3}H - 3.36	0.997**
	% Cl = 0.96 x % ^{3}H - 1.21	0.989*
	% N = 0.90 x % ^{3}H - 1.20	0.999***
Pukepuke	% K = 0.90 x % ^{3}H + 1.83	0.978*
	% Cl = 0.97 x % ^{3}H + 1.32	0.999***
	% N = 0.82 x % ^{3}H + 1.24	0.998**

(a)Fast draining cores



(b)Slow draining cores

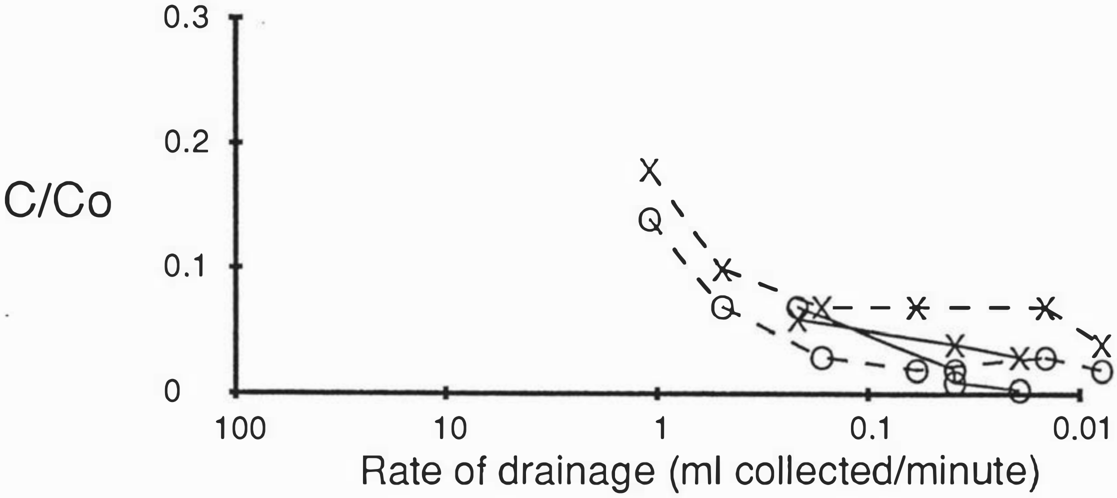


Figure 6.12 The effect of drainage on the relative activity of tritium (X) and concentration of K^+ (O) in the leachate from fast and slow draining cores. Only data for the Pukepuke (—) and Egmont (- -) soil types are presented.

The lack of K adsorption by the soil during the preferential flow of urine through intact soil cores contrasts with the results from experiments using columns of repacked, sieved soil. In the latter, K is not able to bypass the soil matrix via soil macropores and in such situations the soil chemical properties are important in reducing the leaching of K (Davies et al., 1962). In this experiment using intact soil cores the soil properties, CEC and the amount of exchangeable K^+, were of little value in predicting the proportion of K leached from the urine application. Instead the amount of urine K (and also the other ions) leached was related to the rate of preferential flow that occurred after application.

6.4.2 Preferential Flow of Simulated Rain Events

Preferential flow of applied rain water down soil macropores can be initiated when rainfall intensities are in the order of 1 to 10 mm hour^{-1} (Beven and Germann, 1982). Since the application rate of the simulated rainfall in this experiment was greater than 10 mm hour^{-1}, it may be expected that the rain water could have moved preferentially down the macropores particularly of the fast draining cores. The use of tritium as a tracer in the final rain event confirms that preferential flow of the simulated rain events did occur.

In the fast draining cores the relative activity of tritium in the leachate was lower in the leachate collected from the final rain application than that in the leachate collected immediately following the urine application (Table 6.7). Such a result suggests that the flow of rain water through the soil cores was less preferential than the urine flow. Reasons for the lower preferential flow from the rain events include the lower rate of rainfall application (0.08 mm s^{-1}).

6.4.3 Potassium Leached from Simulated Rain Events

During the 48 hours immediately following the urine application, the urine K that remained in the soil had the opportunity to diffuse into the soil matrix and exchange with the soil surfaces. Balasubramanian et al. (1973), Francis (1986) and White et al. (1986) have shown that this reduces the susceptibility of the solutes to subsequent leaching.

Preferential flow of the simulated rain events will reduce solute loss by limiting the interaction between the rain water and the relatively immobile soil water into which the solutes had diffused. Hence, when the rain events began (Figures 6.4, 6.5; Tables 6.3, 6.4; Appendices IV, V) there were immediate decreases in the activity of tritium and the concentrations and amounts of K^+, Cl^- and NH_4^+-N in the leachates for the fast draining cores. In contrast, for the slow draining cores, the flow of rain water through the soil core was slower with a greater opportunity for mixing with or displacement of the resident soil water. Consequently there were higher concentrations and amounts of K^+, Cl^- and NH_4^+-N, and higher activities of tritium in the leachates from the simulated rain events (Figures 6.4, 6.5; Tables 6.3, 6.4; Appendices IV, V).

The difference in the rate of preferential flow of the simulated rain water between the fast and slow draining cores is also reflected in the amounts of exchangeable soil K^+ remaining at the end of the experiment. After application of the urine to the soil, the majority of the K that was retained by the soil as exchangeable K^+ could be found in the top of the soil profile (Figure 3.3). In the fast draining cores the applied rain water tended to bypass most of the soil matrix. This left the majority of the urine K in the top 0-5 cm depth of soil at the end of the experiment (Figure 6.7). In contrast, in the slow draining cores, a greater proportion of the urine K had moved down the soil core and was found as exchangeable K^+ in the 5-10 and 10-15 cm depths (Figure 6.7). The greater retention of urine K within the top 5 cm of soil in the fast draining cores was similar to the results obtained by Wild and Babiker (1976) and Shuford et al. (1977) from field trials on soils at field capacity using NO_3^- and Cl as tracers. They found that the 50-180 mm of irrigation water applied to the soil bypassed the majority of the NO_3^- and Cl in the 0-15 cm depth of soil due to rapid preferential flow of the irrigation water through the soil profile.

During the simulated rain events, a smaller proportion of the applied urine K was leached from all cores compared with the proportion of the tritium that was leached (Tables 6.3, 6.4). Such an effect is predictable as K is strongly adsorbed by the negatively charged soil particles in all these soil types while tritium does not interact with soil particles (Corey and Horton, 1968). However, it appears that adsorption of urine K by the soil will not occur to any noticeable extent until soil water flow rates are low enough for the K to exchange with the soil surfaces. Further evidence for this comes from the effect of the rate of drainage on the relative activity of tritium and relative concentration of K^+ in the leachate collected from the urine application (Figure 6.12). When the drainage was slow the relative K^+ concentration in the leachate was less than the relative tritium activity, but where drainage was fast there were no differences between the relative K^+ concentration and relative tritium activity. If simulated rainfall had commenced immediately after urine application, before the K was able to diffuse away from the immediate vicinity of macropores into the soil matrix, the leaching losses may have been higher.

Except for one of the Egmont cores (core 4), the fast draining cores appear to have leached similar proportions of the applied urine K regardless of soil type (3-6%; Table 6.4). For the slow draining cores there was more K leached from the Egmont and Pukepuke soils (12-27%) compared with that leached from the Tokomaru and Manawatu soils (7-15%; Table 6.4). Since no data were collected on the amounts of native K leached from the cores in this experiment, it is difficult to interpret what effect the native soil K status had on the amount of K leached. It is probable that a greater proportion of the native K was leached from the slow draining cores compared with the fast draining cores because there was evidence of greater native soil water displacement in these slow draining cores. Overall these results indicate that the physical characteristics of the soil have a major effect on K leaching through the influence of the extent to which rain water interacts with the K in the soil core.

6.4.4 Preferential Loss of Nitrogen and Other Elements Following a Urine Application

Preferential flow of urine through the soil following a urination event resulted in a loss of N as well as K (Table 6.5; Figure 6.11b). Up to 62% of the N applied in the urine was lost in preferential flow in this experiment (Table 6.5). As for K, this flow beyond the plant rooting depth represents a major loss of N from the grazed dairy pasture. Despite the large number of publications on the fate of urine N applied to soils, losses of N due to preferential flow of urine have not been considered before (section 5.3.2.1).

Preferential flow of urine will also result in losses of S from grazed pasture as approximately 50% of the ingested S by dairy cows is excreted in urine (Wilkinson and Lowrey, 1973). Phosphorus losses due to preferential flow of urine will be minimal as dairy cows excrete most P in the form of dung not urine (Hutton et al., 1967).

6.4.5 Extrapolation of Results to the Field Situation

The results from this experiment showed that between 0.03 and 74% of the K applied in urine was leached preferentially through the soil. If a typical cow excretes 85 kg K yr^{-1} in urine (section 2.4) and of this, 76 kg K yr^{-1} is returned to the paddock (assuming that 10% is deposited in unproductive areas; section 2.7.3.2), then 2-56 kg K cow^{-1} will be lost per annum in preferential flow of urine beyond the 15 cm depth of soil. For a farm with a stocking rate of 3 cows ha^{-1} this equates to a loss of 6-168 kg K ha^{-1}.

Similarly the amount of K leached by the simulated rain events was equivalent to between 3 and 27% of the K applied in urine. If all the K was leached by the rain events was derived from urine then for a farm with a stocking rate of 3 cows ha^{-1}, each of which deposits 76 kg K yr^{-1} to the paddock, between 6 and 60 kg K ha^{-1} yr^{-1} will be lost in drainage from the rain. Data published in the literature shows that leaching losses of 13-21 kg K ha^{-1} yr^{-1} have been measured for dairy farms (Campkin, 1985). However, Campkin's data may underestimate the amount of K leached from pasture as his data are based on measurements made from water collected from drains, and it is possible that some of the K leached down the soil profile beyond the 15 cm depth does not arrive in the drains but is adsorbed by the soil further down the profile.

The amounts of K leached from the soil cores in this experiment during preferential flow of urine and in drainage from the simulated rainfall events may not resemble K leaching losses in the field. For instance, the amount of urine and rain water which flowed vertically through the soil may be greater from the cores compared with field soil if the cylinder walls prevent conduction of liquids along lateral macropores (Ritchie et al., 1972). In the field, flow of water below the 15 cm depth of soil would be determined by saturated hydraulic conductivity of the surrounding soil, particularly in the lower profile. Detaching the soil at the 15 cm depth could alter the rate of flow of water out of the bottom of the core. This principle has been demonstrated by Bouma et al. (1976). These effects can be reduced by using larger undisturbed cores with a minimum diameter of 30 cm (Bouma and Dekker, 1978; Bouma, 1980; 1982), but there are then limitations to the number of cores and treatments that can be incorporated into experiments conducted in controlled-climate chambers.

The whole experiment described in this chapter was conducted over a 6 to 8 week time period which was a considerably shorter time span than the May to October period being simulated. However, since the objectives of the experiment were related to the leaching of K and not related to the plant recovery of urine K which had previously been investigated (Chapters 3 and 5), it was considered that the time span could be reduced to 6-8 weeks. In hindsight, whereas the short time period may have had no effect on the immediate preferential loss of urine K, the amount of K leached from the simulated rain events may have been overestimated. The reasons for this overestimation were that plant uptake of urine K was reduced by the short experimental period, and that the short time interval between the rain events may have accentuated leaching from the slow draining cores.

In this experiment with intact soil cores, the urine and simulated rain ponded on the slow draining cores until it had soaked into the soil. However, in the field the applied liquids would have been able to move sideways through surface run off thus covering a larger surface area of soil and so providing a greater opportunity for finding a preferential pathway into and through the soil. It is possible that sites in the field represented by these slow and fast draining cores rarely exist, and that the amount of urine K that is leached through the soil profile in the field is somewhere in between the extremes shown in this experiment. Therefore measurements of the amount of urine K that are leached need to be made in the field. Urination events in the field are 2000 ml in size (Table 2.4) not 200 ml as used in this experiment and so plot sizes in the field will be larger and less variable.

6.5 CONCLUSIONS

Preferential flow of tritium labelled urine through soil macropores occurred in all of the intact soil cores used in this experiment. As a result of preferential flow up to 74% of the urine K and 62% of the urine N moved through the soil cores. Although not measured, a similar proportion (60-70%) of the S in the urine would also have been expected to be leached. These results suggest that preferential flow of urine beyond the 15 cm soil depth could result in significant losses of K, N and S from the grazed dairy pasture.

There appeared to be no difference between the four soil types in the amount of urine K that flowed preferentially through the soil despite the different chemical properties of these soils. Instead the major influence on the extent to which preferential flow occurred was the rate at which the urine could move through the soil, with greatest losses occurring when the first 25 g of leachate were collected within 100 s of the time of urine application. At these flow rates there was little opportunity for a chemical reaction to occur between the soil and the solutes in the urine.

In contrast to the proportion (<74%) of urine K that was leached preferentially immediately after application, a smaller proportion (<27%) of urine K was leached from the simulated rain events. The amount of urine K leached from rain events is likely to be influenced by several factors. For example, where drainage was rapid the simulated rain water moved preferentially through the cores which resulted in little interaction between rain and the urine K that had been retained by the soil. Consequently leaching losses were small (only 3-6% of the applied urine K). The longer urine K is retained in the soil the greater the movement into micropores, the greater the adsorption by soil surfaces and the lower the leaching losses of K caused by macropore flow.

In the slow draining cores the high retention of NH_4^+ and a lack of nitrification (possibly caused by a very high soil pH or anaerobic conditions) resulted in an increase in soil pH. As demonstrated in Chapter 4 this increased pH may affect the CEC of the soil.

The drainage rates and amounts of K lost from both the urine application and the simulated rainfall events measured in this experiment were extremely variable. These variations may be less in the field situation where larger areas of soil are covered during urination events by dairy cows.

In this experiment the greatest losses of urine K occurred during the preferential flow of urine beyond a soil depth of 15 cm, which suggests that this mechanism has the most potential for creating losses of K from grazed dairy systems. The next chapter measures the losses of K that occur due to preferential flow of urine under field conditions.

CHAPTER 7

PREFERENTIAL MOVEMENT OF DAIRY COW URINE FOLLOWING APPLICATION TO PASTURE UNDER FIELD CONDITIONS

7.1 INTRODUCTION

In the preceding chapters (Chapters 5 and 6) it has been clearly demonstrated that, immediately after deposition, significant proportions of dairy cow urine can move through the top 15 cm of the soil profile by preferential flow. Furthermore, this preferential movement of urine will result in large losses of K from the soil-plant-animal cycle because little of the K that moved beyond 15 cm was recovered by plant roots (Chapter 3). These losses of K due to preferential flow were measured using intact soil cores (15 cm in diameter and depth) both in the field (Chapter 3) and removed from the field (Chapters 5 and 6). Extrapolation of these results to the larger field situation is not always realistic due to a restriction of lateral flow in the 15 cm diameter soil cores (Ritchie et al., 1972), and, in the case of the experiments in Chapters 5 and 6, there was flow out of the bottom of the cores at the 15 cm depth (Bouma et al., 1976). Obviously in order to quantify the amount of urine K lost preferentially from a grazed dairy pasture it is necessary to measure the preferential loss *in situ* and this was the major objective of this chapter.

Preferential flow of water through soil macropores can occur under the influence of gravity and is initiated when the application rate exceeds the rate at which water infiltrates vertically and horizontally into the soil matrix (Beven and Germann, 1982). Therefore the amount of dairy cow urine which flows preferentially through the 0-15 cm depth of soil will depend partly on the physical characteristics of the soil and the soil moisture content (Bouma, 1981). Another factor that may affect the preferential loss is the soil surface microtopography which could channel the urine into contact with, or away from, the soil's macropores (Beven and Germann, 1982). In the experiment using intact cores collected from the field (Chapter 6) there appeared to be little difference in the amount of urine K that moved preferentially between the four soil types used. However, in the field these soils have contrasting drainage properties to each other (Table 7.1) and different amounts of urine K could be lost in preferential flow.

In the field, preferential flow through macropores can best be characterised by following the fate of tracer solutes and dyes because standard pedological and morphological descriptions of soil structure have been found to be inadequate (Bouma, 1981). Ideally in an

Table 7.1 Some properties of the soils used in the first field experiment. Details of soil classification, bulk density, site location etc. are given in Tables 5.1 and 5.2.

Soil name	Texture	Physical properties of 0-15 cm depth of soil
Tokomaru	silt loam	-well structured soil containing planar cracks at low moisture content -poorly drained during winter months
Manawatu	fine sandy loam	-moderately structured soil -free draining
Egmont	loam	-well structured friable soil -free draining
Pukepuke	sandy loam	-poorly structured soil -despite the presence of shallow drains, over the winter period the water table is near to the soil surface -soil surface water repellent at low moisture content

experiment designed to measure K losses from dairy pastures, K should be used as the tracer. However, tracing the fate of the applied K would be difficult because of the large amounts of indigenous soil exchangeable K^+ (up to 300 kg K ha^{-1} to a 15 cm depth) and the large spatial variation in exchangeable K^+ that exists in many grazed dairy pastures. Results from an earlier experiment showed that during preferential flow of dairy cow urine through four different soil types, similar proportions of the K^+, N, Cl^- and tritiated water (^3H_2O) applied in urine were leached (Figure 6.11). Each of these ions differs in its reaction with soil surfaces from strongly adsorbed K^+ to non-reactive tritiated water, therefore it seems that the preferential flow of the urine occurred too quickly for there to be any significant reaction between the ions moving through the soil and the surrounding soil surfaces (section 6.4.1). This result suggests that any tracer that will move with the urine could be used to measure the amount of K that moves rapidly beyond the 15 cm depth of the profile.

Water soluble dyes such as methylene blue, rhodamine, fluorescein and pyramine have proved useful in tracing water movement through the soil *in situ* (e.g., Reynolds, 1966; Kissel et al., 1973; Bouma and Dekker, 1978; Omoti and Wild, 1979; Scotter and Kanchanasut, 1981; Trudgill et al., 1983). These dyes are adsorbed by the clay and organic matter particles in the soil which makes them useful for identifying areas where preferential flow has occurred, but limits their role in a quantitative description of the value of flow.

Most of the published quantitative research on field leaching has been involved with predicting movement of NO_3^- through the soil profile. Tracers for this have included tritiated water, Cl^- and bromide (Br^-) (e.g., Cameron et al., 1979; Omoti and Wild, 1979; Richter and Jury, 1986)

In the experiments reported in this chapter Br (potassium bromide; KBr) was chosen as a tracer for urine. Background concentrations of Br^- were very low in the soils used in this study, and the methods for the extraction and measurement of Br^- concentration were quick and simple to carry out, hence it was practical to handle relatively large numbers of samples. In addition, Br^- has been successfully used before as a tracer in a leaching study on one of the soil types (Tokomaru silt loam) used in this experiment (Kanchanasut and Scotter, 1982). Methylene blue dye was also used as a visual tracer to observe the occurrence of preferential flow. These observations were used to interpret the distribution of the Br^- ion in each soil profile.

7.1.1 Objectives

The objectives of this field experiment were to:

i. determine the proportion of urine K that was retained in the top 15 cm of *in situ* soil profiles following a simulated dairy cow urination event.

ii. measure whether different soil physical properties and soil moisture contents influenced the proportion of urine that was retained in the upper 15 cm of the soil profile.

iii. determine the variation in the amounts of urine retained within and across different field soils.

7.2 MATERIALS AND METHODS

7.2.1 The Simulated Urine Event

Dairy cow urinations were simulated by applying KBr solution (20.05 g dm^{-3}) to five areas of pasture chosen at random at sites considered to be representative of a particular soil type. The K^+ concentration was similar to that in the urine described in Table 6.1. The KBr solution (2 dm^3) was poured on to the pasture at a rate of 0.2 dm^3 per second from a plastic bottle held 1 m above the ground. This equates to the volume of a typical dairy cow urination (Davies et al., 1962) applied at the rate recorded by Goodall (1951).

A 1 m x 1 m frame was placed over the ground area covered by the simulated urination. This frame supported a 10 x 10 grid marked out with string to give 100 squares, each 7.5 cm wide. Collection of soil samples began 20 minutes after the simulated urination which allowed time for preferential movement of the KBr through the soil profile. Results from an earlier experiment (Chapter 6) showed that preferential flow of applied urine occurred within 20 minutes from the time of application. One soil sample (2 cm in diameter x 15 cm in depth) was collected from each of the 100 squares of the grid using a soil corer (2 cm internal diameter; Figure 7.1). The Br^- content of each soil sample was measured separately (section 7.2.2.1). Extra cores were collected from around the outside of the grid to determine the gravimetric moisture content of the soil (section 7.2.4).

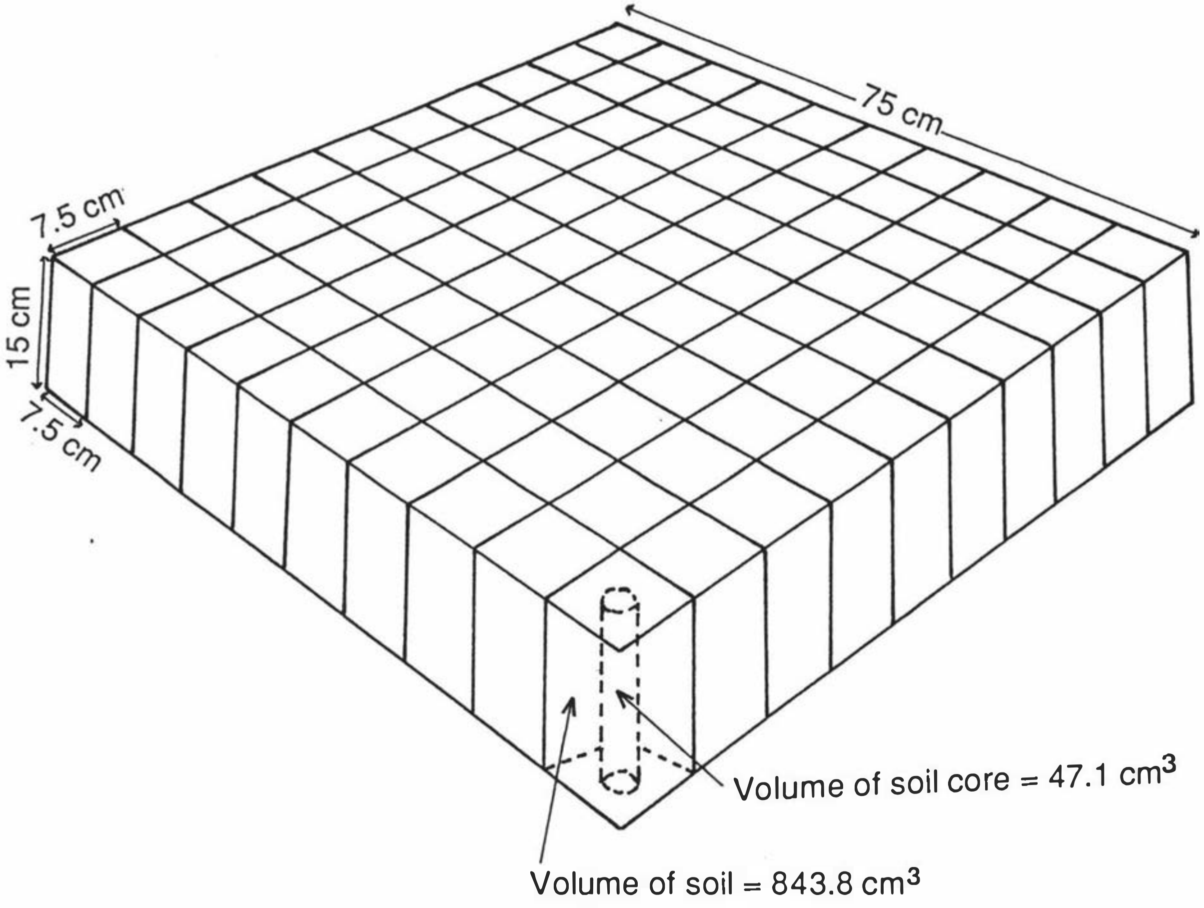


Figure 7.1 A diagrammatic representation of the grid used to collect soil cores and the volume of the cores in relation to the volume of soil represented by each core.

An application of dye was also carried out at each site. In a similar manner to the KBr, 2 dm^3 of 0.1% methylene blue dye (tetramethylthionine chloride) were applied to the soil surface. After 20 minutes had elapsed, the soil profile beneath the application was exposed and the dye pathways observed.

7.2.2 Measurement of Bromide and Recovery From Soil

7.2.2.1 *Bromide analysis*

The Br^- was extracted from each soil core in distilled water using a soil : water ratio of 1 : 5. Sufficient 5M sodium nitrate (as an ionic strength adjustor) was added to provide a 2% solution (Abdalla and Lear, 1975). The samples were stirred thoroughly and left at room temperature overnight. Then they were stirred again and left until the soil had settled to the bottom of the container. The Br^- concentration in the solution was measured directly using an Orion solid state Br^- electrode and a reference electrode connected to an Orion 901 Digital Ionalyser. No background concentrations of Br^- were detected in any of the soils.

The herbage on each soil core was not removed but included in the sample for extraction. This would have recovered any Br^- that had adhered to the herbage during the application of the KBr solution. The short time scale of this experiment is unlikely to have resulted in significant amounts of Br^- being taken up by the plants.

7.2.2.2 *Evaluation of soil adsorption of the bromide*

The ability to quantitatively recover the applied Br was investigated on the Tokomaru and Egmont soils in a laboratory experiment. These two soils were chosen for their contrasting anion sorption capacities and organic matter contents (Table 5.1).

Three replicates comprising 10 g of either Tokomaru or Egmont soil were wetted to give moisture contents of 0.22 g g^{-1} and 0.43 g g^{-1} respectively. These moisture contents were the same as when the sites were sampled in the summer (Table 7.2). The moist samples were incubated at room temperature with 1 ml of 0.1M KBr for 24 hours. The extractable Br^- content of each soil sample was determined by the method described in section 7.2.2.1.

Similar Br^- recovery tests were also carried out on the Taupo and Rukuhia soils by spiking some of the soil cores (with 5 ml of 0.1 M KBr) after their initial Br^- concentration had been determined. After the addition of the spike the samples were stirred, left to stand for 4 hours, stirred again and the final Br^- concentration measured.

Table 7.2 The soil gravimetric moisture content at time of sampling (g g^{-1}).

Soil		Winter	Summer
Tokomaru		0.35	0.21
	SEM	0.007	0.012
Manawatu		0.34	0.22
	SEM	0.003	0.022
Egmont		0.65	0.43
	SEM	0.025	0.006
Pukepuke		0.49	0.32
	SEM	0.012	0.028

7.2.3 **Field Experiments**

7.2.3.1 *First experiment*

In the first experiment the four soil sites described in Tables 5.1, 5.2 and 7.1 were used. The experimental procedure described in section 7.2.1 was conducted twice, once in summer of 1987 (March) when the moisture content was drier than field capacity (Table 7.2) and again in the winter of 1987 (May) when the soil was at field capacity (Table 7.2).

7.2.3.2 *Second experiment*

In the second experiment the same procedure as described in section 7.2.1 was carried out on three more soil types commonly used for dairy production in New Zealand. These three soils were from the Waikato region and included a yellow-brown loam (Waihou sandy loam) from a different geographical area to the Egmont soil, a yellow-brown pumice soil (Taupo sandy loam) and an organic soil (Rukuhia peat). The latter two soils contrast with the other soil types as they have different textural and structural properties yet are also important dairying soils in New Zealand. As before, for each soil type a site was selected on an established dairy pasture. Some information on the physical properties and the moisture contents at the time of sampling of these sites is presented in Table 7.3.

For this experiment the sampling was only carried out once in December 1987. Since the moisture contents at the time of sampling were quite high (0.41-0.65 g g^{-1}) these sites have been described as "winter" sites for the purposes of comparison with the soils from the first experiment. As discussed in section 7.2.7 the number of replicates was increased from 5 to 10 on the Taupo and Rukuhia soils. To reduce the number of analyses required for 10 replicates, the 100 individual cores were bulked together at 5 of the replicate sites on the Taupo and Rukuhia soils. The same soil : water ratio and concentration of ionic strength adjustor, as described in section 7.2.2.1, were used for Br^- measurement.

7.2.4 **Gravimetric Water Content Analysis**

The moisture content of the soil was measured as the weight loss due to drying at 105^oC for 16 hours and expressed as the weight of water : dry weight of the soil.

Table 7.3 Some properties of the soils used in the second field experiment.

	Soil name Waihou	 Taupo	 Rukuhia
N.Z. soil group	yellow-brown loam	yellow-brown pumice	organic
Soil taxonomic classification	Umbric Dystrandept	Typic Vitrandept	Typic Mediasaprist Histosol
Texture	sandy loam	sandy loam	peat
Location from Hamilton	40 km East	55 km South/East	20 km South
Map reference	T14 490870	T16 467358	S15 195608
Bulk density (Mg m^{-3})	0.8	0.7	0.6
Gravimetric water content (g g^{-1})	0.65	0.41	0.57
Physical properties	well structured free draining	well structured free draining	fibrous topsoil artificially drained

7.2.5 **Statistical Analysis**

For each soil in the first experiment the mean Br recoveries for summer and winter were compared using the *t* test (Steel and Torrie, 1960). An analysis of variance was also carried out on the data on Br recovery from the winter sampling on all sites.

7.2.6 **Method Evaluation**

7.2.6.1 *Bromide recovery from soil*

Between 100 and 107% of the applied Br^- was recovered in water extracts of the Tokomaru and Egmont soil samples (Table 7.4). Bromide adsorption by soil has also been shown to be minimal in the Tokomaru silt loam (Kanchanasut and Scotter, 1982) and in a Californian alluvial loamy sand (Abdalla and Lear, 1975).

7.2.6.2 *Coring technique for field experiments*

The percentage recovery of the applied Br^- in the field experiments was determined by collecting 100 soil cores from a 75 cm x 75 cm area. Each soil core had a volume of 47.1 cm^3 and the concentration of Br^- in this core was considered to represent a soil volume of 843.8 cm^3 (Figure 7.1). A calculation based on using the 47.1 cm^3 core to represent a soil volume of 1687.6 cm^3 (i.e., 50 cores per 75 cm x 75 cm area rather than 100 cores) showed that the estimate of the percentage of Br^- recovered would have been similar (Table 7.5). A further calculation based on taking only 25 cores from the 75 cm x 75 cm area also shows a similar estimate of Br^- recovery to that based on 100 cores (Table 7.5). Therefore there seems to be no advantage in taking more than 100 cores to predict the Br^- recovery in the top 15 cm of the soil profile.

7.2.7 **Variability of Field Observations**

Using the data from the first field experiment and Stein's two-stage equation (Steel and Torrie, 1960) it was possible to calculate the number of replicates required to achieve a reasonable estimate of Br^- retention for each site. An acceptable coefficient of variation (CV) for this type of experiment was considered to be 20%, based on the data from previous experiments with Br^- as a tracer on the Tokomaru silt loam by Kanchanasut (1980). To obtain an estimate within 10% of the true mean and with a CV of 20%, then 4 replicates were required. Hence the 5 replicates per site used in the first field experiment provided an

Table 7.4 The recovery of applied Br from 4 soils after laboratory incubation.

Soil	% Br recovered	Mean	SEM
Tokomaru	101, 101, 104	102.0	1.00
Egmont	101, 103, 106	103.3	1.45
Taupo	102, 103, 107	104.0	1.53
Rukuhia	100, 102, 106	102.7	1.76

Table 7.5 The effect of the number of cores (25, 50 or 100) taken from the same 75 cm x 75 cm area on the estimate of Br retention in field soils.

Number of cores	% Br^- retained	SEM
100	60	0.2
50	63	0.6
25	65	1.1

acceptable compromise between improving the accuracy of an estimate and experimental effort. The one exception was the Manawatu soil in the winter which exhibited a high degree of variability at this time of sampling (CV of 30%; Appendix VI)

The data from the first experiment on the Egmont soil indicated that 5 replicates would be adequate for characterising preferential flow on the second yellow-brown loam soil sampled. In the absence of such information for the Taupo and Rukuhia soils, 10 experimental replicates were examined at each of these sites.

7.3 RESULTS AND DISCUSSION

7.3.1 Effect of Soil Type and Season on Bromide Recovery

The recovery of Br^- from individual field sites across different soils ranged from 39 to 113% (Appendix VI). The mean recoveries ranged from 52% (Taupo) to 106% (Pukepuke, summer; Figure 7.2; Appendix VI). The analyses described in section 7.2.6.1 show that if the Br^- is present in the soil it can be recovered and measured, therefore preferential movement of Br^- below 15 cm is the major reason for the low recoveries of Br^-. Observations on the movement of the methylene blue dye at each site support this viewpoint (Figures 7.3 - 7.6).

There was little difference in Br^- recovery between the two seasons for the Tokomaru soil (Figure 7.2). There appears to be a difference between the two seasons on the Manawatu soil. However, because of the high variability in Br^- recovery particularly at the winter sampling, this difference was not significant (Figure 7.2). The dye study in both of these soils clearly showed that preferential movement to depths greater than 15 cm below the soil surface occurred along the numerous channels created by plant roots and earthworms (Figure 7.3). In the Tokomaru soil there was also considerable movement through soil fracture planes (Figure 7.4). Other observations on the Tokomaru silt loam have shown that methylene blue dye can move through a combination of these types of macropores to mole drains situated 40 cm below the soil surface (Scotter and Kanchanasut, 1981).

Application of dye to the Egmont soil also showed that preferential flow down root and earthworm channels could occur to depths greater than 15 cm. This site contained a population of grassgrubs (*Costelytra zealandica*) and there was some movement of the dye down the channels created by the burrowing activity of these larvae (Figure 7.5).

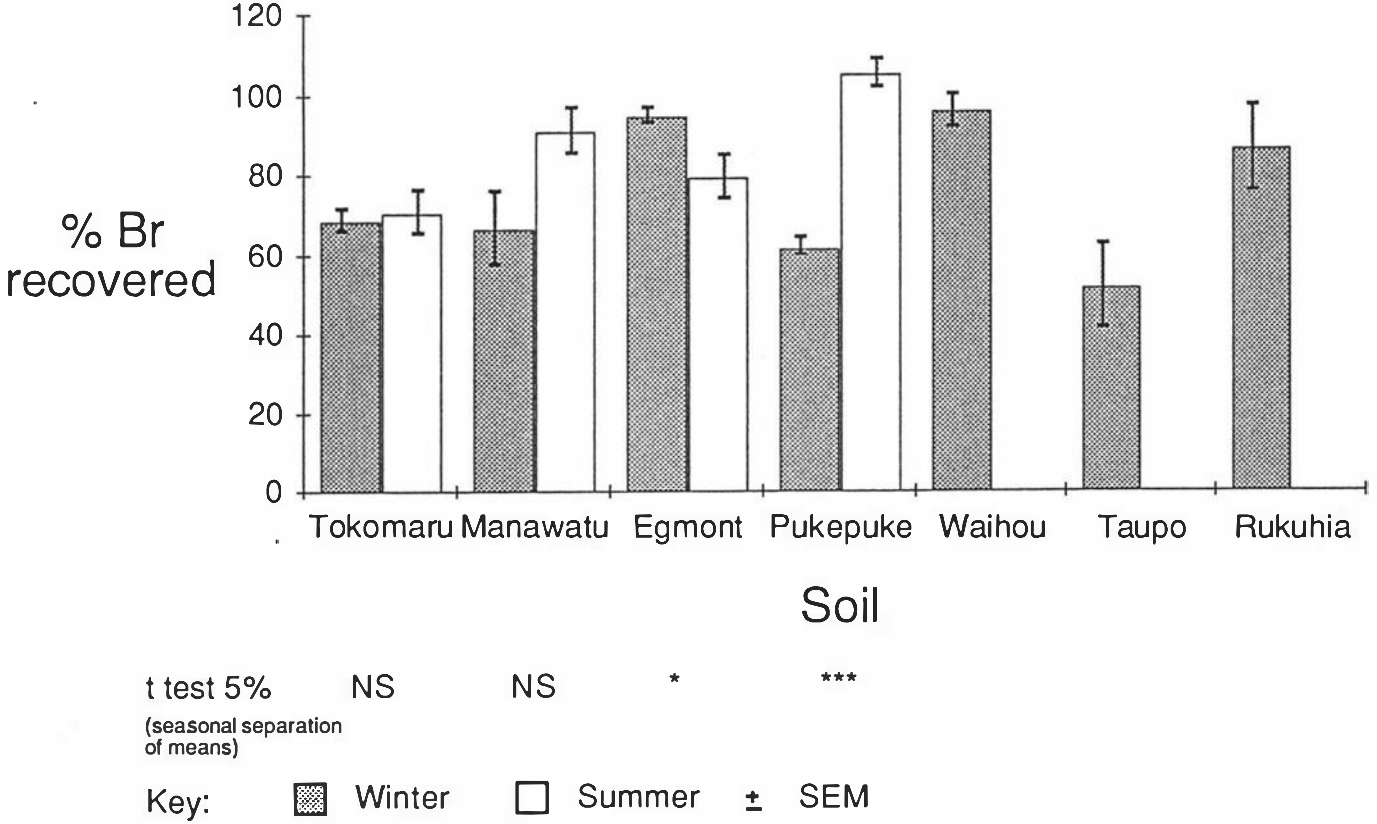


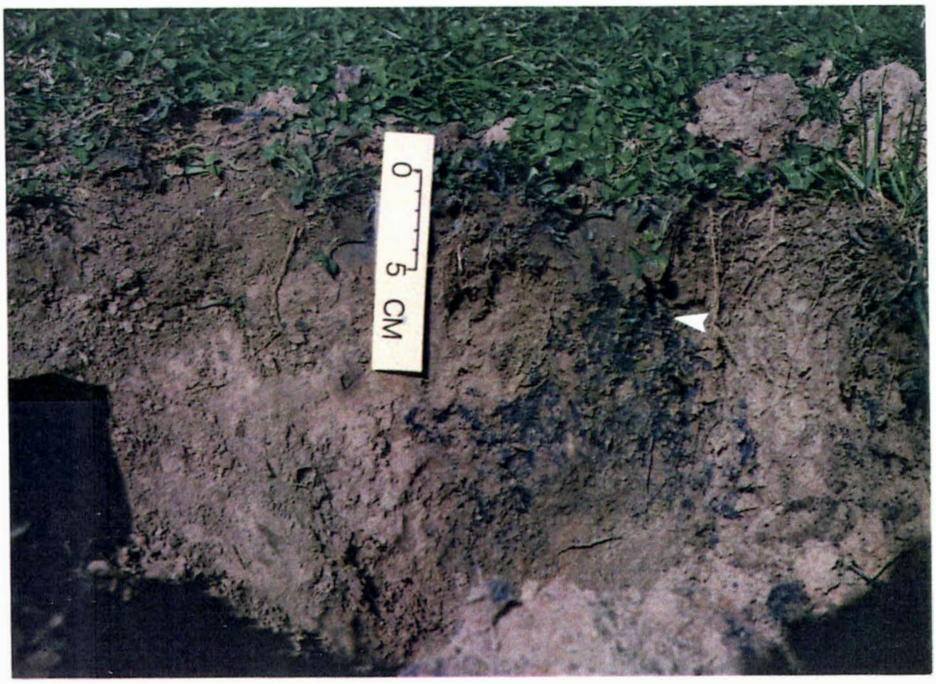
Figure 7.2 The mean recovery of Br from the 0-15 cm depth of seven soils after urine deposition on pasture was simulated using KBr solution.

◄ Figure 7.3

The distribution of methylene blue dye through soil macropores created by plant roots (◂) and earthworms (◄) in the Manawatu silt loam following a simulated urination event of 2 dm^3 of dye applied to the soil surface in May 1987.

Figure 7.4 ▼

The distribution of methylene blue dye through soil macropores created by soil fracture planes (◂) in the Tokomaru silt loam following a simulated urination event of 2 dm^3 of dye applied to the soil surface in March 1987.



◄ Figure 7.5

The distribution of methylene blue dye through soil macropores created by plant roots (◄) and grassgrub channels (◄) in the Egmont brown loam following a simulated urination event of 2 dm^3 of dye applied to the soil surface in May 1987.

Figure 7.6 ▼

The distribution of methylene blue dye through soil macropores created by plant roots (◄) in the Pukepuke black sandy loam following a simulated urination event of 2 dm^3 of dye applied to the soil surface in March 1987.

The Egmont soil showed a higher retention of Br^- in the winter (95%) compared with the summer (80%) (Figure 7.2). The KBr ponded on the soil surface during the winter experiment indicating that the surface may have been compacted during the last grazing. This effect would limit the infiltration of the KBr. Movement of solute below a restrictive layer such as this tends to be nonpreferential (Kanchanasut and Scotter, 1982) and hence more Br^- was retained. Such a result suggests that if a soil type has a tendency to become compacted by a grazing animal, particularly when the soil has a high moisture content, then preferential losses of urine K may be minimal under these conditions.

On the Pukepuke soil (which has marked seasonal hydrophobic properties; M. Wallis, pers. comm.) there appeared to have been very little movement of the dye below 15 cm in the summer. Some movement of the dye down the profile was observed around plant roots but this did not extend beyond 5-7 cm below the soil surface (Figure 7.6). The dark blue dye ponded on the surface for up to 15 minutes before it penetrated the Pukepuke soil surface (at both the summer and winter sampling times) and dye may have been adsorbed by the soil organic matter during this time. This would have limited the potential of the dye as a tracer further down the profile particularly since the darker colour of this soil made it difficult to see areas of low dye intensity. In the summer the KBr tended to pond on the soil surface until it infiltrated into the soil. Since there was little movement of the Br^- below 15 cm (Figure 7.2), presumably the water repellent nature of the soil surface restricted infiltration to a rate at which water was mainly absorbed and did not flow through the soil.

In contrast there was a significantly lower recovery of Br^- in the winter at the Pukepuke site (Figure 7.2) suggesting that there was some movement of the Br^- beyond a depth of 15 cm. Extra soil samples were collected from around the outside of the grid, which was placed over each simulated urination, to check for lateral movement of KBr but no evidence for this was found. The accompanying dye study did not highlight any areas where preferential flow may have occurred but as already discussed observations of the dye movement may have been of limited value in interpreting Br^- retention at this site.

The rate of preferential flow down through a soil profile can be influenced by the soil moisture content (Bouma, 1981). Preferential flow is more likely to occur when the soil is at field capacity, with a reduction in preferential flow at moisture contents lower than this (Hoogmoed and Bouma, 1980; Scotter and Kanchanasut, 1981). The data from the Pukepuke soil appears to follow this pattern.

7.3.2 Retention of Bromide on Other Soil Types

The majority of the applied Br^- (96%) was retained in the Waihou soil (Figure 7.2). As for the other yellow-brown loam soil (Egmont) at a similar moisture content, the KBr tended to pond on the surface before infiltration. The accompanying dye study showed little preferential movement through the profile. There was some movement through earthworm channels but this tended to be lateral rather than vertical.

In comparison with the Waihou soil, retention of Br^- was lower in the Rukuhia (87%) and Taupo (52%) soils (Figure 7.2), indicating that preferential flow was occurring to a much greater extent in these soils, especially in the Taupo soil. As discussed in section 7.3.1 for the Pukepuke soil, observations of preferential flow through the soil profile in the Rukuhia and Taupo soils using methylene blue dye were made difficult by the high organic matter content of the top layer of soil which adsorbed the dye as it penetrated into the soil below.

7.3.3 Effect of the Distribution of the Potassium Bromide Solution Over the Soil Surface on the Amount of Bromide Recovered

Within each soil and at each sampling time there was a range of Br^- retention values. A correlation of Br^- retention in the top 15 cm of soil with the area of soil covered by the applied KBr (Figure 7.7) shows that if the KBr could not move rapidly into and through the 15 cm depth of soil it tended to pond and spread across the soil surface. Reasons for these differences in infiltration ability could be due to variations in vegetation cover, and surface microtopography caused by the presence of large pores open to the surface and depressions caused by animal treading. Observations made at the time of the experiment showed that where the KBr did not immediately penetrate the soil it was channeled via these surface irregularities. This effect is similar to that reported by Cameron et al. (1979) who found that soil surface microtopography influenced the redistribution of applied water across the soil surface and resulted in extremely variable patterns of Cl^- and NO_3^- leaching through the soil profile.

The effect of the physical condition of the soil surface on Br^- retention is clearly illustrated by the Egmont soil (Figure 7.7). This soil shows a lower retention (80%) when infiltration was rapid and therefore the area covered was small (0.16 m^2). However, during the winter when the soil surface was compacted by grazing animals the KBr ponded on the surface (area = 0.21 m^2) and the retention in the top 15 cm of soil was high (95%).

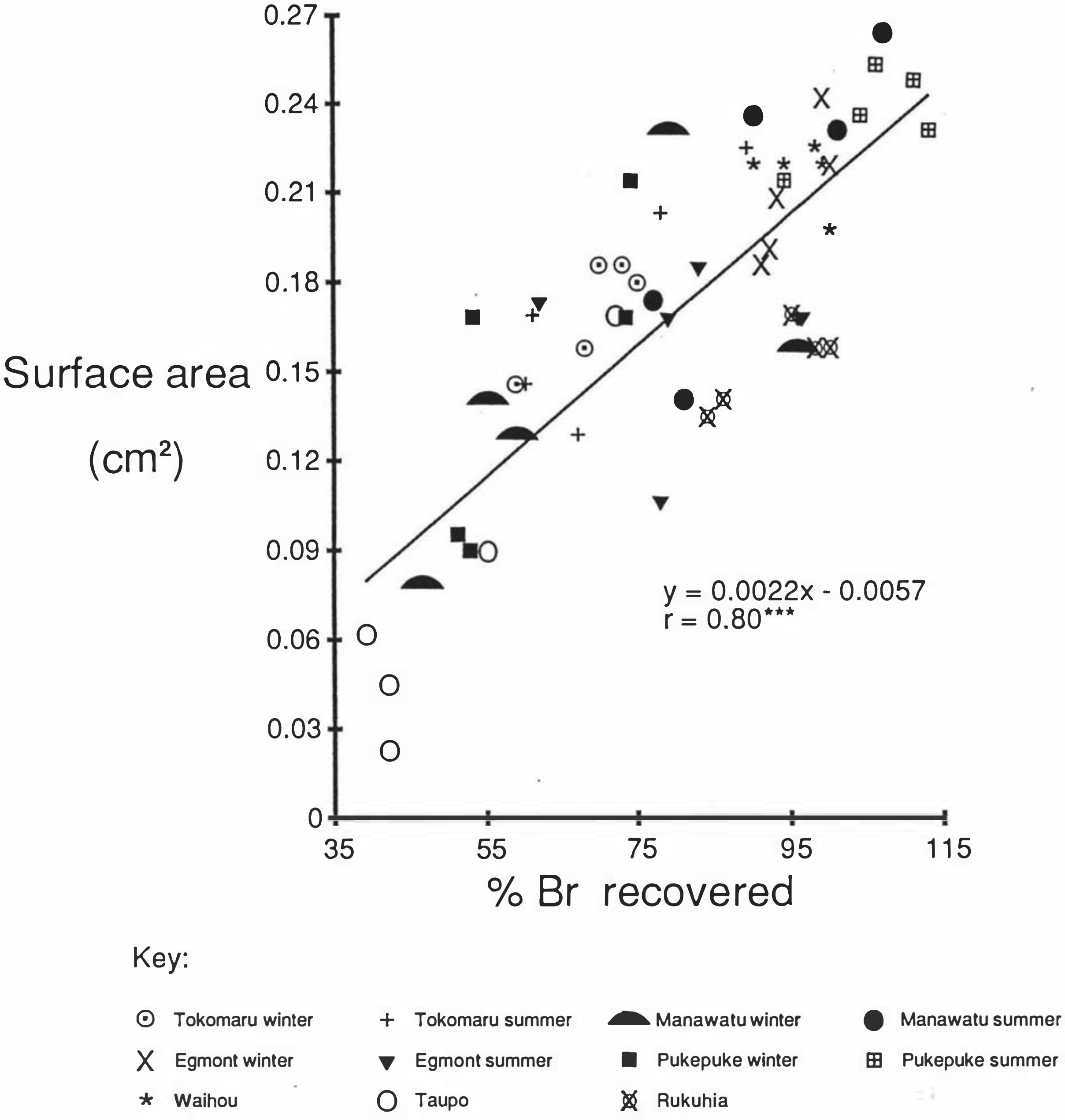


Figure 7.7 Relationship between area covered by the applied KBr and Br recovery (%) in the 0-15 cm soil depth of seven soils.

These results suggest that the soil surface conditions (e.g., microtopography, water repellency and treading damage) were an important factor in determining Br^- retention. While there were statistical differences between some of the soil types (e.g., Br^- recovery from the Taupo soil (52%) was significantly less than the recovery from the Pukepuke summer (106%), Waihou (96%), Egmont winter (95%) and Rukuhia (87%) soils; LSD 0.05%; Figure 7.2) within replicates of most soils there was a high degree of variation in Br^- recoveries as shown in Figure 7.7.

Sixty four percent of the variation ($r^2 = 0.64$) in Br^- retentions has been explained by their linear relationship with the surface area covered by the KBr application (Figure 7.7). Use of a quadratic equation to describe this relationship gives a similar correlation coefficient and so offers no advantage over the linear equation. Other factors which may have influenced Br^- retention but which were not measured in this experiment were the density of pasture plants and the plant species present. Pasture plants can cause variability in leaching patterns of applied solutions due to interception and stem flow (Kanchanasut and Scotter, 1982).

7.4 GENERAL DISCUSSION

The relationship between soil surface area wetted and retention of Br^- may provide a quicker method of estimating preferential leaching losses than the technique used in this experiment which involved collecting and analysing 100 soil cores for each replicate. Instead it may be possible to measure the area covered by a tracer on the ground and use this to predict the retention of the tracer in the top 15 cm of soil. Measurement of the ground area could be rapidly carried out using methods such as infra red photography and a heat producing tracer such as hot water. This technique could be used to predict preferential losses of urine on other soils which have different surface microtopographical and physical properties (e.g., drainage characteristics) to the soils used in this experiment. One example is the gley podzol soils which have massive structure and impeded drainage throughout the soil profile; physical properties unlike any of the soils tested in these experiments.

The data presented in Figure 7.7 show that the KBr applications (simulated urine events) covered a surface area of between 0.02 and 0.26 m^2. Very few measurements of the surface area covered by dairy cow urinations have been published apart from 0.20 m^2 (Davies et al., 1962) and 0.28 m^2 (Petersen et al., 1956a) for one soil type in New Zealand and North Carolina respectively. Other figures ranging from 0.13 to 3.52 m^2 are reported by Hogg (1968), Richards (1973, cited in Morton, 1984), Wilkinson and Lowrey (1973), and Richards and Wolton (1976), but it is unclear whether some of these values are measurements of actual areas covered by urine or assessments of the pasture area exhibiting growth responses to the urine. Measurements based on the area of pasture visibly affected by urine tend to be larger than the surface area wetted by the urine (Lotero et al., 1966). The Br^- recovery data presented in Figure 7.7 suggests that the range of wetted areas observed in this study and by other authors may represent a range of urine K losses from the rooting depth of pasture. This range of urine K losses also suggest that the K fertiliser requirements will vary between these sites.

The effect of season on the surface area covered by a urination event has been investigated previously by Blagden (1969) in England using simulated applications comprising 3.5 dm^3 of cow urine. The results of this research showed that during the summer (June to August) the area covered by the urine ranged from 0.5 to 1.0 m^2, while in the spring and autumn the area increased to between 0.9 to 2.2 m^2. Unfortunately details of the soil type and moisture content and the subsurface flow of urine were not presented but these results are similar to those recorded for the Egmont soil (Figure 7.7) with a greater area covered by KBr at the higher moisture content.

7.5 CONCLUSIONS

A technique has been developed which uses KBr solution to estimate the amount of urine that would be retained in the top 15 cm of the soil profile within 20 minutes of application. In the simulated urine applications the unrecovered urine (KBr) moved beyond this depth due to preferential flow through a network of macropores such as soil cracks, earthworm and plant root channels.

Results from this technique showed that between 52 and 106% of the applied Br^- was retained within the top 15 cm of the soil profile for the seven soils, which contrasted in their physical and chemical properties. The different amounts of Br^- that were retained in each soil varied with the season. This variation in Br^- retention did not appear to directly relate to differences in soil moisture or soil type (with the possible exception of the Pukepuke site). Instead the retention of Br^- correlated well with the extent to which the simulated urine application (KBr) was distributed over the soil surface. This varied according to soil surface conditions such as the microtopography, water repellency and treading damage by the grazing animals. It may also vary with slope, type of vegetation (e.g., prostrate or upright species, density of plants) and the presence of a water table near the soil surface.

A more rapid method of estimating the preferential loss of urine at a particular site would be to measure the surface area of ground covered by a simulated urination event. Then from the relationship between the area of soil surface wetted and retention of Br^- the amount of urine K (and urine N and S) that would be lost preferentially at a particular site could be calculated.

The movement of up to 48% of urine beyond the 15 cm plant root depth will result in considerable losses of K from the soil-plant-animal cycle. The information from this and previous chapters can be used to estimate the amount of K in dairy cow urine that will be lost preferentially beyond the 15 cm soil depth from farms on these soil types. Such information can be combined with data describing other losses of K that occur from dairy farms to provide an estimate of K fertiliser requirements as shown in Chapter 8.

CHAPTER 8

MASS BALANCE MODELLING OF POTASSIUM LOSSES FROM THREE GRAZED DAIRY FARMS

8.1 INTRODUCTION

Simple models capable of estimating the nutrient requirements of any farming system can be constructed using a mass balance approach. These models are based on the assumption that the flows of nutrients through the plant available pool within the system are in steady state and plant and/or animal nutrient requirements can be maintained if nutrient losses from the system are equalled by nutrient gains. Such an approach is currently used by the New Zealand Ministry of Agriculture and Fisheries (MAF) in their Computerised Fertiliser Advisory Service (CFAS; Comforth and Sinclair, 1984), which is used to estimate the fertiliser requirements for individual farms.

The CFAS K model has been described in detail in section 2.8. The main principles are that losses of K from the cycling pool occur through the removal of K from the farm in animal products (milk, calves and culled cows) and the transfer of excretal K to the unproductive areas of the farm such as the dairy shed, raceways and feedpads. These losses caused directly by the dairy cow are known collectively as the animal loss factor (ALF). Further K loss occurs through leaching of K below the effective plant root uptake depth. Gains of plant available K to the cycling pool originate from inputs of fertiliser K, purchase of supplementary feed from outside the farm, and the release of K from the nonexchangeable and structural forms of K in the soil.

The review of literature (Chapter 2) has shown that whereas the principles for modelling K gains and losses are well known, no attempts appear to have been made to test the accuracy of the CFAS model. The main reason for this lack of appraisal has been the scarcity of suitable data from which to both estimate and measure the gains and losses for a given system. Losses of K via leaching are particularly difficult to estimate.

Data obtained from the experiments reported in Chapters 3, 5, 6, and 7 quantitatively describe the short term (one grazing interval) and long term (one growing season) fate of K from urine deposited on grazed pasture. These data are used in this chapter to estimate the losses of K that occur from grazed dairy pastures and to construct mass balance models describing the annual K losses and gains for three seasonal supply dairy farms.

8.1.1 Objectives

The objectives of this study were to:

i. predict the losses of K that occur from grazed dairy pasture by using a mass balance model with estimates of K loss in animal products, K transfer to unproductive areas and K leaching, and to compare these losses with changes in soil K over time.

ii. compare the actual K losses and the estimates of K losses calculated in (i) with the losses predicted by the CFAS model, and to examine critically areas where the two models differ.

8.1.2 General Site Information

The three farms used were a farmlet on the Taranaki Agricultural Research Station, situated near Normanby in South Taranaki on an Egmont brown loam soil. The second site was the commercial dairy farm of J.D. & R.D. Wallace on Rukuhia peat near Hamilton, and the third site was the Massey University No. 4 dairy farm on the Tokomaru silt loam soil type. These farms were the locations of experiments reported in previous chapters. As in the preceding chapters, the boundary condition for K lost by leaching was taken as the 15 cm soil depth, because the zone of maximum root activity by pasture plants was above this depth (Chapter 3). At these sites attempts were made to compare the estimates of K losses from these farms with actual measurements of changes in soil K. For the No. 4 dairy farm it was possible to include in the calculation of gains and losses an estimate of the amount of urine K that was subsequently taken up by the pasture from the 0-15 cm depth of soil during the growing season, thus providing information on the recycling of K through the compartments of the grazed pasture.

8.2 TARANAKI AGRICULTURAL RESEARCH STATION

8.2.1 Materials and Methods

The data used in the model for the Taranaki Agricultural Research Station were collected over an eight year period from one of the farmlets involved in an evaluation of the effects of grass grub on dairy production. The size of the farmlet, number of Friesian-Jersey crossbred cows grazed and relevant production data is presented in Table 8.1. Weekly pasture growth was measured by visual estimation. Each visual estimate was corrected to dry matter production ha^{-1} using a regression equation which described the relationship between visual estimates and dry matter yield. Pasture consumed was measured as the difference between pre-grazing and post-grazing herbage mass for each paddock in the farmlet. Every January, soil samples (0-7.5 cm) were collected from each paddock for K analysis by the MAF Quick test (section 2.6.1.4). Fertiliser was applied annually in either August or September.

Young stock were reared off the farmlet and all feed supplements used were derived from the unit. Details of the soil type (Egmont brown loam) and site are given in Tables 5.1, 5.2 and 7.1.

8.2.2 Results and Discussion

8.2.2.1 *Estimation of potassium losses from animal products*

The amount of K lost in milk production (Table 8.2) is estimated from the weight of milk produced and a mean K content of 0.13% (Hutton et al., 1967; reviewed in section 2.7.3.1).

On average, each cow spent 4 years in the farmlet's herd. Since the mean cow liveweight was 400 kg, this amounted to a loss of 100 kg per year for cow replacement. During these 4 years the cow produced 3 calves that were culled (the fourth calf was kept as the cow's replacement), each at a liveweight of 25 kg. Using this information the amount of K lost annually in culled cows and calves was calculated (Equation 2, Appendix VIIa) at 0.2-0.3 kg K su^{-1} (Table 8.2).

Adding these two components together gives a mean annual animal production loss of 0.65 kg K su^{-1} (Table 8.2). Using data for a typical dairy farm in the Waikato, Middleton and Smith (1978) estimated a similar loss of 0.61 kg K su^{-1}.

An alternative method of estimating the K loss in milk production was to calculate the amount of K ingested and assume that 5% (a value used by Hutton et al. (1967) and adopted in the CFAS model) was retained for milk production. Intake was calculated as the difference in herbage mass between pre- and post-grazing measurements made for each paddock at each grazing. Using this method the animal production loss was 1.01 kg K su^{-1} (Table 8.3).

Table 8.1 Production data from the farmlet on the Taranaki Agricultural Research Station.

Season	Farmlet size (ha)	No. of cows	Stock units[1] per cow (su cow^{-1})	Milkfat production (kg ha^{-1})	Pasture growth (kg DM ha^{-1})
1978/79	11	32	6.6	480	9630
1979/80	10	33	7.2	600	12120
1980/81	12	44	6.6	607	15350
1981/82	6.48	24	6.4	540	13620
1982/83	6.48	24	7.5	723	17800
1983/84	6.48	24	7.7	737	19680
1984/85	6.48	23	7.4	640	16880
1985/86	6.48	24	8.0	816	16170

[1] from Cornforth and Sinclair (1984)

Table 8.2 Estimates of the amount of K removed in animal products calculated from the production data (Table 8.1) on the Taranaki Agricultural Research Station.

Season	Milk production (kg ha^{-1})	K in milk[1] (kg K su^{-1})	K in calf and[2] cow replacement (kg K su^{-1})	Total K removed in animal products (kg K su^{-1})
1978/79	9553	0.65	.03	0.68
1979/80	12071	0.66	.03	0.69
1980/81	11925	0.63	.03	0.66
1981/82	10075	0.55	.03	0.58
1982/83	13435	0.63	.02	0.65
1983/84	14153	0.65	.02	0.67
1984/85	12308	0.61	.02	0.63
1985/86	14356	0.63	.02	0.65
			Mean	0.65

[1] Assume milk contains 0.13% K (Hutton et al., 1967)

[2] Assume 18.25 kg calf @ 0.17% K and 100 kg cow replacement @ 0.15% K (Middleton and Smith, 1978)

Table 8.3 Estimates of the amount of K removed in animal products calculated from the pasture intake data on the Taranaki Agricultural Research Station.

Season	Pasture consumed ($kg\ DM\ su^{-1}$)	K consumed[1] ($kg\ K\ su^{-1}$)	K in milk[2] ($kg\ K\ su^{-1}$)	Total K in[3] animal products ($kg\ K\ su^{-1}$)
1978/79	528	18.1	0.91	0.94
1979/80	512	17.5	0.88	0.91
1980/81	559	19.1	0.96	0.99
1981/82	551	18.8	0.94	0.97
1982/83	599	20.5	1.03	1.05
1983/84	667	22.8	1.14	1.16
1984/85	636	21.8	1.09	1.11
1985/86	547	18.7	0.94	0.96
			Mean	1.01

[1] A mean value of K was used of 3.42%. This was obtained from samples collected from each paddock at two different times in one of the later years of the trial. This K content is believed to be typical of the herbage content for the whole 8 years.

[2] Assume 5% of ingested K is retained for milk production (Hutton et al., 1967)

[3] Assume that the K in calf and cow replacement is the same as in Table 8.2

Three reasons may explain this higher value:

Firstly, the values calculated in Table 8.3 were based on 5% of the K ingested being used for milk production, whereas for this farmlet the actual amount of K in milk (Table 8.2) represented only 4% of the K consumed by the dairy cows (Table 8.3).

Secondly, there were difficulties in accurately measuring the quantity of herbage consumed by dairy cows. The data on K consumption in Table 8.3 have been derived indirectly from assessments of pre- and post-grazing herbage mass and show that the cows utilised between 89 and 105% of the pasture grown [utilisation = pasture consumed su^{-1} (Table 8.3) x su ha^{-1} (Table 8.1) / pasture growth ha^{-1} (Table 8.1)]. Such utilisations are very high (some are obviously overestimates) and are probably higher than on a commercial farm.

Thirdly, the K content of the pasture was not analysed at each grazing and so the mean K content of 3.42% may represent another inaccuracy in the calculation. The K loss from the measured production data (Table 8.2) represents the actual K lost in animal products, regardless of how much pasture K was utilised. It is therefore more realistic to use the loss of 0.65 kg K su^{-1}, calculated in this manner, in the subsequent calculations.

8.2.2.2 *Estimation of losses from transfer of excretal potassium to unproductive areas*

Over the whole milking season the cows on the farmlet spent an average of 45 minutes in the dairy shed and raceways at each milking. For two milkings this equates to 6% of the day. Assuming that the amount of excreta deposited in these areas was proportional to the time spent there, then 6% of the excreta was deposited in these unproductive areas and so was lost from the soil-plant-animal cycle. The mean milking season on the farmlet was from mid-August to mid-May, and for the remaining 3 months of the year the cows remained on the paddocks for 24 hours a day. Therefore, over a whole 12 month period, 5% of the excreta can be considered to have been deposited off the paddock. The K in this excreta amounted to 0.91 kg K su^{-1} (Table 8.4).

Commercial properties have larger herd sizes than this farmlet, and their cows may spend a longer period of time off the paddock. In these situations the first cow may return within 45 minutes and the last cow within 2 hours. This equates to a mean of 2.8 hours off the paddock per day during lactation, and 10% of the excreta being deposited in the dairy shed and raceways for a 12 month period. Hancock (1950), Goodall (1951), Cooper (in Steele, 1982) and During (1984) report similar estimates of 2-8%, 11%, 12% and 10-15% respectively, for excreta transfer to unproductive areas. Their data also tend to support the assumption made earlier that the amount of excreta deposited in the dairy shed and raceways is

Table 8.4 Estimates of the amount of K lost through deposition of excreta off the paddock in the dairy shed and raceways on the Taranaki Agricultural Research Station.

Season	K consumed (kg K su^{-1})	K excreted[1] (kg K su^{-1})	K deposited[2] off the paddock (kg K su^{-1})	K deposited[3] off the paddock (kg K su^{-1})
1978/79	18.1	16.7	0.84	1.67
1979/80	17.5	16.1	0.81	1.61
1980/81	19.1	17.6	0.88	1.76
1981/82	18.8	17.3	0.87	1.73
1982/83	20.5	18.9	0.95	1.89
1983/84	22.8	21.0	1.05	2.10
1984/85	21.8	20.1	1.01	2.01
1985/86	18.7	17.2	0.86	1.72
		Mean	0.91	1.81

[1] Assume 92% of ingested K is excreted (Hutton et al., 1967)
[2] Assume 5% of excreted K is deposited off the paddock
[3] Assume 10% of excreted K is deposited off the paddock

proportional to the time the cows spend there. Assuming that 10%, rather than 5%, of the excreta is deposited off the paddock, the loss is increased from 0.91 to 1.81 kg K su^{-1} (Table 8.4) which may be more appropriate on commercial farms.

8.2.2.3 *Estimated animal loss factor*

Combining the K lost for milk and calf production, cow replacement, and transfer of excretal K off the paddock results in a total ALF of 1.6 kg K su^{-1}. This value is considerably less than the 4 kg K su^{-1} currently used in CFAS to predict K requirements for easy-rolling land (Cornforth and Sinclair, 1984). Campkin (1985) derived the ALF of 4 kg K su^{-1} from the following: 0.8 kg K su^{-1} due to loss in milk and calf production and cow replacement plus 3.2 kg K su^{-1} due to transfer of excretal K to the dairy shed and raceways. While the CFAS value for the amount of K lost in animal products (0.8 kg K su^{-1}) is similar to the value calculated for the Taranaki Agricultural Research Station (0.65 kg K su^{-1}), the CFAS transfer loss is considerably higher than the 0.9 kg K su^{-1} calculated for the farmlet in this study. Campkin (1985) provides no direct experimental evidence for using 3.2 kg K ha^{-1} as a transfer loss. His value was derived from different sources in the literature, which included countries other than New Zealand, therefore, conclusions drawn by Campkin (1985) for the New Zealand situation must be tentative. The K losses calculated for the farmlet in this study are based on actual observations and so form a reliable measurement of the time spent in the dairy shed and raceways.

8.2.2.4 *Estimation of losses of potassium due to leaching*

Leaching losses of K below the effective plant rooting depth can be due to both preferential flow of urine after application to the soil and leaching due to rainfall. The latter is estimated from the annual drainage per year (calculated as the sum of the amounts of rainfall that exceed evapotranspiration each month with an allowance for soil water storage) and a K concentration in the drainage water of 4 $\mu g\ ml^{-1}$ (Cornforth and Sinclair, 1984). This equates to an annual loss of 11 kg K ha^{-1} or between 71 and 121 kg K per farmlet (Table 8.5).

The mean loss of urine K due to preferential flow beyond a soil depth of 15 cm at this site was measured as 12% (Figure 7.2). This will have resulted in a loss ranging from 272 to 526 kg K per annum for the farmlet over the eight years (Table 8.5).

While the leaching losses have been partitioned into two categories (rainfall and preferential flow of urine) and each has been calculated separately, some of the K in the drainage water may have been from preferential flow of urine through the soil into the drains.

Table 8.5 Estimates of the amount of K leached from preferential movement of dairy cow urine after deposition on the paddock, and from rainfall on the Taranaki Agricultural Research Station.

Season	K deposited[1] on the paddock (kg K su^{-1})	K leached[2] preferentially from urinations (kg K su^{-1})	(kg K $farmlet^{-1}$)	K leached in rainfall (kg K $farmlet^{-1}$)
1978/79	15.9	1.71	361	121
1979/80	15.3	1.65	392	110
1980/81	16.7	1.81	526	132
1981/82	16.4	1.77	272	71
1982/83	18.0	1.94	349	71
1983/84	20.0	2.15	397	71
1984/85	19.1	2.06	351	71
1985/86	16.3	1.76	338	71

[1] From Table 8.4

[2] Assuming 90% of K deposited on paddock is urine (Hutton et al., 1967) and 12% is lost due to preferential movement through the soil profile (refer to Figure 7.2)

Conversely, not all of the K that is leached beyond the pasture plant rooting depth makes its way into the drainage water. Thus the sum of the two leaching parameters (preferential flow and drainage) may overestimate the amount of urine K lost preferentially and underestimate the amount of K leached from rain events.

8.2.2.5 *Comparison of the estimated changes in the net balance of potassium in the cycling pool with actual changes in soil potassium*

A comparison of the losses of K from the farmlet, as estimated by the ALF, with the amount of fertiliser K applied is shown in Table 8.6. The losses of K were calculated with both an ALF of 1.6 kg K su^{-1} as derived in section 8.2.2.3, and the CFAS ALF of 4 kg K su^{-1}. The results show that when the ALF of 4 kg K su^{-1} is used there is a net loss of K in every year, while the ALF of 1.6 kg K su^{-1} results in a net gain. Other gains and losses of K to the farmlet may occur through the release of K from sources of nonexchangeable soil K^{+} and leaching. The release of nonexchangeable K^{+} on this soil type has not been measured, but the amount released is thought to be low (Metson, 1980).

Leaching losses of K due to preferential flow of urine beyond plant root depth can be substantial from this farmlet (Table 8.5) and so need to be included. When the leaching losses (Table 8.5) are combined with the ALF of 1.6 kg K su^{-1} (Table 8.6), the total annual loss for the farmlet exceeds the gain from fertiliser by up to 376 kg K per farmlet (Table 8.7) or 50 kg K ha^{-1} yr^{-1}. If the leaching losses are used in conjunction with an ALF of 4 kg K su^{-1} the net loss of K per year is significantly higher.

It could be expected that the difference between losses and gains in soil K (<50 kg K ha^{-1} yr^{-1}; Table 8.7) would result in a decline in available soil K. However, over the eight year period of this study there was no indication that soil K in the 0-7.5 cm depth decreased as measured by the QTK (Table 8.8). If the change in soil K content of 50 kg K ha^{-1} yr^{-1} was uniform with depth in the 0-15 cm depth of soil, then the QTK in the 0-7.5 cm depth could be expected to have dropped by 2 QTK units yr^{-1} (Cornforth and Sinclair, 1984). A decline in QTK of 2 units yr^{-1} is unlikely to be detected because of the variability in QTK values (expected error per value is 30%; I.S. Cornforth, pers. comm.). Over the eight year period the accumulated losses of K (230 kg K ha^{-1}; Table 8.7) could have resulted in a drop in 10 QTK units. The fact that a change of this magnitude was not observed (Table 8.8) indicates that the soil exchangeable pool as measured by QTK, was being replenished by release of K from other sources in the soil or that actual K losses were smaller than predicted.

Table 8.6 The predicted amount of annual animal K loss (ALF) for the whole farmlet using ALF factors of both 1.6 and 4.0 kg K su^{-1}, and the amount of annual K gain due to fertiliser application on the Taranaki Agricultural Research Station.

Season	Annual animal loss ALF = 1.6 (kg K $farmlet^{-1}$)	 ALF = 4.0 (kg K $farmlet^{-1}$)	Annual gain (fertiliser K) (kg K $farmlet^{-1}$)
1978/79	336	840	800
1979/80	381	952	700
1980/81	468	1171	750
1981/82	246	614	482
1982/83	288	720	465
1983/84	296	739	443
1984/85	273	682	513
1985/86	307	767	405

Table 8.7 The predicted total amount of K lost for the whole farmlet from both animal production and leaching using an annual animal loss factor (ALF) of 1.6 (for animal losses due to production and transfer) and the amount of annual K gain due to fertiliser application on the Taranaki Agricultural Research Station (kg K $farmlet^{-1}$).

Season	Animal loss	Leaching loss preferential	 rainfall	Total loss	Annual gain (Fertiliser K)
1978/79	336	361	121	818	800
1979/80	381	392	110	883	700
1980/81	468	526	132	1126	750
1981/82	246	272	71	589	482
1982/83	288	349	71	708	465
1983/84	296	397	71	764	443
1984/85	273	351	71	695	513
1985/86	307	338	71	716	405

Table 8.8 MAF Quick test K results for the Taranaki Agricultural Research Station (mean for all paddocks on the farmlet; MAF Quick test units; 0-7.5cm depth of soil).

Season	Quick test K
1978/79	12
1979/80	15
1980/81	10
1981/82	10
1982/83	13
1983/84	13
1984/85	12
1985/86	15

Other measurements of soil K, such as total K, may not be as susceptible to the variability of the QTK. However, the total amount of native K in most New Zealand soil types is very high (Table 2.1), which renders detection of comparatively small changes in soil K due to dairy farming extremely difficult. It would, for instance, be impossible to measure the decrease of 50 kg K ha^{-1} yr^{-1} or 230 kg K ha^{-1} per 8 years at the Taranaki Agricultural Research Station against the background K levels of 20,000 kg K ha^{-1} in the 0-15 cm depth of soil (Metson, 1980). Because of this difficulty it was decided to repeat the comparison of estimates of K loss with actual changes in soil K using an organic soil which has a low native K content (approximately 80 kg K ha^{-1} in the 0-15 cm depth of soil; New Zealand Soil Bureau, 1968). This will be discussed in section 8.3.

Using the CFAS model the fertiliser requirement for the Taranaki Agricultural Research Station was calculated at 14 kg K ha^{-1} yr^{-1} (Appendix VIII). As already discussed the CFAS model predicted much larger losses of animal K than were calculated in this study (Table 8.6). In addition the CFAS model predicted that the rate of soil supply would have been 105 kg K ha^{-1} yr^{-1} (Appendix VIII). If the K losses are equivalent to those estimated in this study and soil exchangeable K levels are being maintained (as they seem to be), then it appears that the rate of supply of K was at best 50 kg K ha^{-1} yr^{-1}. Therefore the CFAS K model appears to have overestimated the losses of K from the Taranaki Agricultural Research Station and the rate of K supply from the soil.

8.3 WALLACE FARM

8.3.1 Materials and Methods

In contrast to the research farmlet at Normanby, not all of the data required to construct a K balance was available for the Wallace farm. Data on animal production prior to 1974, pasture yield and herbage K concentration were incomplete, and where necessary best estimates have been made. The K losses were calculated using a mathematical model (the "K Loss" model) which incorporates a series of equations detailed in Appendix VIIa. The calculations were carried out by a personal computer and the software programme "Multiplan". An example of the printout for this site is presented in Appendix VIIc. The losses and gains are expressed on a per hectare basis.

The farm was initially developed in 1955/56 from a deep dome peat bog which is part of the Moanatuatua swamp near Hamilton. A description of the development procedures used to convert this soil type to farmland is given by During (no date). The farm has been under dairying for the 30 years since development. Young stock were reared off the farm and all supplements made were fed back onto the farm. Details of the site are given in Table 7.3.

8.3.1.1 *Measurement of the change in soil potassium status*

Measurements of actual change in soil K over the 30 years were made by comparing the K status of a site on the dairy farm with the closest adjacent area of undeveloped Moanatuatua swamp (1 km away). For the purpose of comparing the K status of the developed and undeveloped sites it was assumed that these sites were pedologically similar, and that over the last 30 years the undeveloped site remained in steady state equilibrium.

During development the surface of the peat bog subsided due to shrinkage as the peat dried out and the organic matter decomposed (Hupkens van der Elst, 1980). In order to measure the K content of the corresponding depth of undeveloped peat that subsided to form the 0-15 cm depth of developed soil, it was necessary to estimate the rate of subsidence for this site. Soil samples were collected down to a depth of 60 cm (the water table depth) in 10 cm increments at the undeveloped site, while at the developed site samples were collected to the 30 cm soil depth in 5 cm increments. Samples were collected from six soil profiles chosen randomly at each site. The amount of mineral matter in each soil depth in the developed and undeveloped sites was measured using the weight loss due to ignition. The iron (Fe) content of each sample was determined (section 8.3.1.2). Soil samples from both sites were also analysed for K and PO_4^{3-}-P content (section 8.3.1.2) and bulk density (section 8.3.1.3).

8.3.1.2 *Soil potassium, iron, phosphate contents and weight loss on ignition*

The soil was oven dried (50°C) and finely ground (<0.1 mm). One g of soil was weighed into a silicon crucible and ashed at 550°C for 1 hour in a muffle furnace. The loss of weight due to ignition was recorded by measuring the weight of the cold sample after it was removed from the furnace. The ash was transferred to a boiling tube with 5 ml of hydrochloric acid (2 parts acid to 1 part water) and digested at 100°C for 4 hours (Beckwith and Little, 1963). When cool, the extract was filtered and diluted to 100 ml. The Fe^{3+} and K^+ concentrations were measured by atomic adsorption (Fe^{3+}) and atomic emission spectroscopy in the presence of 0.2% caesium chloride (K^+). The PO_4^{3-}-P concentration was determined colorimetrically using the vanadomolybdate method of the Association of Official Agricultural Chemists (1975).

8.3.1.3 *Soil bulk density*

The bulk density of each soil depth was determined as the dry weight of soil (oven dried at 105^{o}C for 16 hours) collected in a cylinder 5 cm in diameter and depth. Results are expressed as Mg dry soil m^{-3}.

8.3.2 **Results and Discussion**

8.3.2.1 *Estimation of potassium losses from animal products*

The data available on the stocking rate per hectare and production per cow are given in Table 8.9. Records prior to 1974 were not available as an unrecorded amount of the milk was used for a large scale calf rearing operation. For the period from 1957/58 to 1973/74 a mean annual production has been estimated at 120 kg milkfat per cow, and a stocking rate of 2.4 cows ha^{-1} (based on a mean of the initial stocking rate of 2.1 cows ha^{-1} and a stocking rate of 2.7 cows ha^{-1} in 1974/75).

The amount of K lost from the farm via milk production was estimated from data on milkfat production and a mean annual milkfat content of 4.5% (Equation 1, Appendix VIIa).

The mean cow liveweight was 400 kg and each cow spent 4 years in the herd. Therefore the amount of K lost due to calf production and cow culling (Equation 2, Appendix VIIa) was 0.18 kg K per cow each year.

The total amount of K lost from the farm in these animal products (Equation 3, Appendix VIIa) varied from 8.8 to 16.6 kg K ha^{-1} yr^{-1} (Table 8.9).

8.3.2.2 *Estimation of losses from transfer of excretal potassium to unproductive areas*

No measurements of cow intake or pasture production were available, so the amount of K ingested by the cow was calculated from the milkfat production and the conversion factor of 25 kg dry matter for each kg milkfat (Bryant, 1982). The amount of K ingested per cow was then estimated by assuming that the dry matter ingested had a K content of 3.2%. (The equation used was MF*25*h/100, where MF is the kg milkfat cow^{-1} and h is the %K concentration in the herbage; Appendix VIIa). The 3.2% herbage K concentration is a mean of the data from seven sites in the North Island published by Metson and Saunders (1978).

Table 8.9 Production data, the estimated amount of K lost due to animal production and transfer to unproductive areas, and the estimated amount of urine K lost due to preferential flow of dairy cow urine beyond the effective plant root zone on the Wallace farm.

Season	Stocking rate	Milkfat	Amount of K lost		
			animal production	transfer	preferential flow
	(cows ha^{-1})	(kg cow^{-1})	(kg K ha^{-1})	(kg K ha^{-1})	(kg K ha^{-1})
1957/58-1973/74[1]	2.4	120	8.8	13.7	24.3
1974/75	2.72	127	10.5	16.4	29.2
1975/76	2.72	124	10.2	16.0	28.5
1976/77	2.91	117	10.4	16.2	28.8
1977/78	2.91	134	11.8	18.5	32.9
1978/79	2.91	133	11.7	18.4	32.7
1979/80	3.0	133	12.1	18.9	33.7
1980/81	3.0	136	12.3	19.4	34.5
1981/82	3.6	153	16.6	26.1	46.6
1982/83	3.6	139	15.1	23.8	42.3
1983/84	3.5	127	13.5	21.1	37.6
1984/85	3.5	125	13.3	20.8	37.0
1985/86	3.45	129	13.5	21.1	37.6
1986/87	2.9	141	12.4	19.4	34.6
		Total	312.3	488.3	869.7

[1] Mean values per year

While 3.2% is the best estimate available of herbage K content there may be some error caused by using this value to estimate the total amount of herbage K for all 30 years. For example, if a value of 2.8% (critical value for K in pasture well supplied with N; G.S. Smith et al., 1985) is used instead of the 3.2%, and all the other parameters remain the same, then the total amount of K ingested decreases from 113 to 99 kg K cow^{-1} yr^{-1} which results in a reduction in K losses created by the dairy cow. The significance of a decrease in K intake on the size of the K losses is discussed in section 8.5.1.3.

The K already accounted for in animal products (section 8.3.2.1) was subtracted from the total K ingested and the remainder is the amount of K excreted.

Information on the length of the milking season (260 days) and the time the cows spend in unproductive areas (2 hours per day during the milking season) was available for only one season (1986/87). These values were used as the mean for the past 30 years. Over the year the cows spend 6% of their time in unproductive areas and it is therefore assumed that they deposit 6% of the excreta in the dairy shed and raceways over a 12 month period. The amount of K that was lost due to transfer to unproductive areas (Equation 4, Appendix VIIa) varied from 13.7 to 26.1 kg K ha^{-1} depending on the amount of K ingested each year (Table 8.9).

8.3.2.3 *Estimation of losses of potassium due to preferential flow of urine*

Of the excretal K deposited on the pasture, 90% is considered to be in the form of urine (Hutton et al., 1967). From the measurements carried out on this soil type (section 7.3.2), 13% of the urine deposited on the soil surface is expected to move preferentially beyond the effective plant rooting depth (Figure 7.2). Using Equation 5 in Appendix VIIa, the annual loss of K due to preferential loss of urine was calculated as between 24.3 and 46.6 kg K ha^{-1} (Table 8.9).

8.3.2.4 *Estimation of losses of potassium due to leaching from rainfall*

The amount of K leached per annum from the rainfall events was calculated assuming there were 480 mm of drainage per year (this was estimated from the sum of the amounts of rainfall that exceeded evapotranspiration each month; New Zealand Meteorological Service data) and a K concentration of 4 μg ml^{-1} (Campkin, 1985). This resulted in an annual rainfall-induced drainage loss of 19 kg K ha^{-1}.

8.3.2.5 *Gain in potassium from fertiliser*

During the first two years of development the pasture received heavy applications of 15% potassic superphosphate (Table 8.10). Thereafter maintenance applications of fertiliser were applied, usually as split applications of either 15% or 30% potassic superphosphate in the spring and autumn of each year. Seven years after the initial development the pasture was redeveloped due to compaction of the top 20 cm of soil which had received fertiliser. Pasture roots would not penetrate below this layer, so the soil was recultivated to a depth of 40 cm with lime and potassic superphosphate. This redevelopment resulted in an extra application of 32 kg K ha^{-1} during that year.

8.3.2.6 *Change in soil potassium over the 30 year period*

At both sites, soil bulk density did not change significantly with changes in soil depth. There was, however, a difference in bulk density between the sites, viz., 0.55 Mg m^{-3} (SEM=0.01) for the developed soil and 0.12 Mg m^{-3} (SEM=0.01) for the undeveloped soil.

Ignition and acid digestion of the developed peat, followed by chemical analysis, showed that in the 0-15 cm depth of soil there were 280 kg K ha^{-1} (SEM=56). The largest amount of K was in the samples taken from the 0-5 cm depth and as soil depth increased the soil K content decreased (Figure 8.1). The amount of K in the undeveloped peat was considerably smaller with a total of 48 kg K ha^{-1} (SEM=5) in the 0-60 cm depth (Figure 8.1).

There were also large differences in the amount of Fe and PO_4^{3-}-P present in the soil between the two sites (Figure 8.1). The increased amount of Fe in the 0-15 cm depth of soil at the developed site may have originated from subsidence of a certain original depth of undeveloped peat. By summing the Fe content of each soil depth in the undeveloped peat (Figure 8.1) until the Fe content was equivalent to the Fe content in the 0-15 cm depth of developed peat, it can be calculated that the subsidence rate was >4 m per 30 years. A similar calculation based on the loss on ignition data (Figure 8.1) resulted in a subsidence rate of a similar magnitude. Measurements on the annual rate of subsidence of the Waikato peatlands have shown that the rate is 20 cm per year initially but then it declines to 2 cm per year under pasture (Thompson, 1980). Therefore it is unlikely that the developed peat has subsided at a rate of >4 m per 30 years. A more likely explanation is that the developed peat contained a higher mineral content initially compared with the undeveloped site. The different mineral content of the two sites may have originated through different patterns of deposition of K and Fe from flooding and volcanic ash showers (Green and Lowe, 1985). This suggests that the

Table 8.10 Fertiliser history for the Wallace farm.

Season	kg K ha^{-1}
Development year 1	72
year 2	108
Redevelopment (extra)	32
1957/58-1981/82	[65 kg K ha^{-1} for 25 years] 1625
1982/83	58
1983/84	65
1984/85	58
1985/86	81
1986/87	43
Spring 1987	84
Total	2226

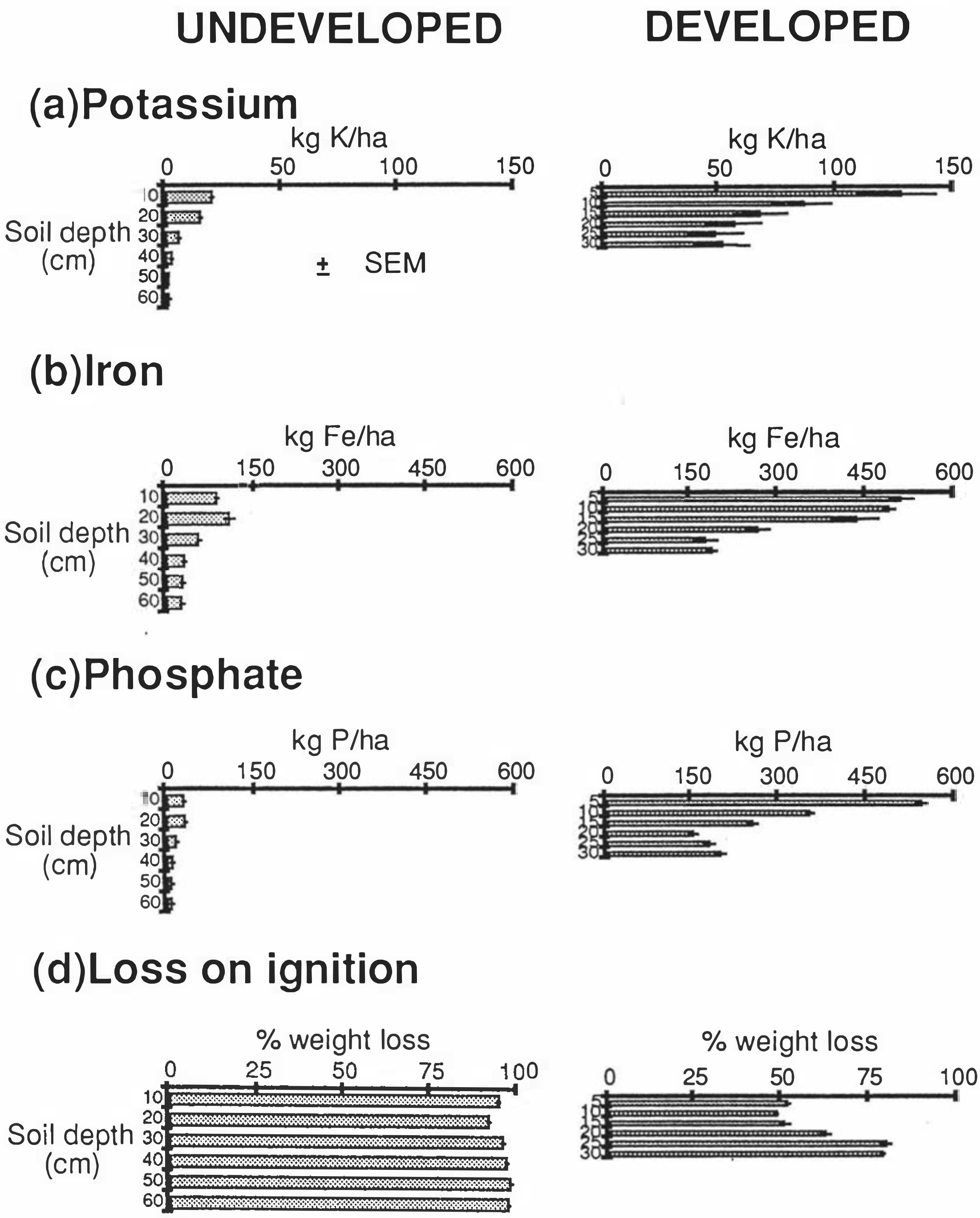


Figure 8.1 The amounts of potassium, iron and phosphate in, and the weight loss on ignition of soil from different depths of undeveloped and developed peat on the Wallace farm.

assumption made in section 8.3.1.1, that the undeveloped and developed sites were initially pedologically similar, is incorrect and that the actual change in K content of the soil to 15 cm depth can only be estimated at being less than or equal to 280 kg K ha^{-1} (SEM=56; the total amount of K present at the developed site). The undeveloped site was the only one available in the area and no better estimate of the change in soil K status can be made.

8.3.2.7 *Comparison of estimated potassium losses with changes in soil potassium content*

When all possible fates for fertiliser K are considered (animal products, transfer to unproductive areas and leaching losses) a balance sheet of the estimated losses and gains of K for the 30 years on the farm shows that the estimated net change in soil K was -14 kg K ha^{-1} (Table 8.11). Therefore it was expected that there would be little change in the soil K status over the 30 years.

The best estimate that could be made of the measured change in soil K showed a gain of <280 kg K ha^{-1} (SEM=56). Obviously there is a difference between the value for the predicted and measured change in soil K (-14 cf <280 kg K ha^{-1}; Table 8.11). This difference is small compared with the total amount of K that has either been lost from the farm (2240 kg K ha^{-1}) or added to the farm in fertiliser (2226 kg K ha^{-1}) over the 30 years. Therefore it can be considered that the parameters used in the K Loss model (Appendix VIIa) provided reasonable estimates of the K losses and gains for this farm.

The CFAS model was also used to predict the losses and gains of K to the farm (Table 8.12). Losses of K in the CFAS model due to animal production and transfer of K to unproductive areas were accounted for by an ALF of 4 kg K su^{-1} and totalled 1986 kg K ha^{-1} for the 30 years (Table 8.12). The total leaching losses from rainfall were estimated to be 318 kg K ha^{-1} (Table 8.12). The model also requires an input for both short and long term supply of K from the soil. However, on this soil type there is no evidence for a significant source of long term K as the total K content is low (280 kg K ha^{-1}) compared with soils where such a source is available (e.g., Tokomaru silt loam which contains 17,000 kg K ha; Metson, 1980). Therefore the soil parameter in the CFAS model (section 2.8) is not appropriate for this soil type and so is not included as a gain of K in the balance sheet (Table 8.12).

The overall balance of gains and losses using the CFAS parameters resulted in a predicted net soil change of -78 kg K ha^{-1} which is close to the result using the K Loss model (-14 kg K ha^{-1}; Table 8.11) developed from observations and experiments described in this

Table 8.11 The predicted amounts of K lost from, and gained to,the Wallace farm, calculated using production data (Table 8.9) and the K Loss model (Appendix VII).

Parameter	kg K ha^{-1}
Losses of K	
Animal Loss	
animal products	312
transfer	488
Leaching Loss	
preferential movement of urine	870
drainage water	570
Total Loss	2240
Gains of K	
Fertiliser	2226
Predicted Net Change in Soil K	-14
Measured Gain in Soil	<280

Table 8.12 The predicted amounts of K lost from, and gained to, the Wallace farm, calculated from the CFAS model. (Details of equations and abbreviations in section 2.8).

Parameter	Equation	kg K ha^{-1}
Losses of K		
Animal Loss		
animal products		
transfer	(SR x ALF) x 30 years. where ALF = 4 kg K su^{-1} SR = 16.55 su ha^{-1}	1986
Leaching Loss	(R - PE) x ((SR/1250) x leaching factor)) x 30 years where R = 1200 mm PE = 800 mm leaching factor = 2	318
Total Loss		2304
Gains of K		
Soil Supply		0
Fertiliser		2226
Predicted Net Change In Soil K		-78
Measured Gain In Soil K		<280

thesis. However, there were differences between these two methods in the way that the K losses were calculated. For the Taranaki Agricultural Research Station (section 8.2.2.3), and the Wallace farm (Table 8.13) the amount of K lost in animal products and in transfer (the ALF) was calculated at 1.6 kg K su^{-1}. In the CFAS model an ALF of 4 kg K su^{-1} gave a considerably larger estimate of the amount of K deposited in the raceways and dairy shed (Table 8.12 c.f. Table 8.11). Whereas the ALF in the CFAS model appears to have overestimated the losses of K due to production and transfer, the CFAS model does not take into account directly the loss due to preferential flow of urine beyond the plant rooting depth (a mechanism that resulted in the greatest loss of K from the farm; Table 8.11). Some attempt is made in the CFAS model to include leaching losses as influenced by stock by modifying the amount of K leached from rainfall with the stocking rate. However, the amount of K leached as calculated in this manner (318 kg K ha^{-1}; Table 8.12) falls short of the amounts of K that were lost by preferential flow and by the rainfall (870 and 570 kg K ha^{-1} respectively, Table 8.11), as calculated by the K Loss model.

It can be concluded that the use of an ALF of 4 kg K su^{-1} in the current CFAS model may give the misleading impression that losses of K could be reduced substantially by decreasing transfer losses, i.e., the time that the cows spend off the paddocks in unproductive areas of the farm. Although the amount of K lost through transfer can be reduced in this manner (e.g., on the Wallace farm if the time is reduced by a total of 30 minutes per day during the milking season the loss is decreased from 1.1 to 0.8 kg K su^{-1}) the potential saving (e.g., 0.3 kg K su^{-1}) is small compared with the amount of K lost through preferential flow of urine (at 1.5 kg K su^{-1}). As discussed in section 8.5.1.1 little can be done to reduce the amount of urine K lost preferentially.

8.4 MASSEY UNIVERSITY No. 4 DAIRY FARM

In addition to calculating the losses of K from the Massey University No. 4 dairy farm by using the K Loss model, a model was also developed to estimate the amounts of K that are transferred between the soil, plant and animal components of the K cycle. It was possible to develop the transfer model for this site because quantitative data was available for the recovery of urine K by pasture plants (Chapter 3). The advantage of the transfer model over the K Loss model is that information is provided on the speed with which K is recycled between compartments within the system, rather than just information on the losses of K from the system. Thus it will be possible to identify situations when the rate at which K is recycled may become a growth limiting factor (such as might occur during periods of rapid plant growth) and therefore becomes an important factor in assessing short term fertiliser requirements.

Table 8.13 Estimation of the animal loss factor due to animal production and transfer of excreta to unproductive areas on the Wallace farm.

Season	Total animal loss[1] (kg K ha^{-1})	Stocking rate[2] (su ha^{-1})	ALF (kg K su^{-1})
1957/58-1973/74[3]	22.5	14.4	1.56
1974/75	26.9	16.6	1.62
1975/76	26.2	16.3	1.61
1976/77	26.6	16.9	1.57
1977/78	30.3	18.0	1.68
1978/79	30.1	18.0	1.67
1979/80	31.0	18.6	1.67
1980/81	31.7	18.9	1.68
1981/82	42.7	23.8	1.79
1982/83	38.9	23.0	1.69
1983/84	34.6	21.0	1.65
1984/85	34.1	21.0	1.62
1985/86	34.6	21.1	1.64
1986/87	31.8	18.6	1.71
		Mean	1.60

[1] Sum of production and transfer losses from Table 8.9
[2] From Cornforth and Sinclair (1984)
[3] Mean values per year

8.4.1 Materials and Methods

The system being considered was defined as one hectare in area, and plant uptake of K was from a soil depth of 15 cm over the time period of one year. The animal production data used in the calculations were for the 1987/88 milking season (1 June 1987 - 31 May 1988). All the necessary data were available either from earlier chapters in this thesis, or previous experiments by other researchers carried out on this soil type (e.g., information on the K lost in drainage) or from other literature (e.g., information on K lost via animal products as detailed in section 8.2.2.1).

During the 1987/88 milking season the stocking rate was 3 cows ha^{-1}. Young stock were reared off the farm and all feed supplements made were fed back on to the property. Over the winter the cows spent 25 days on a feedpad because of the poor drainage properties of the soil type (Tokomaru silt loam; Table 7.1). Further details of the site are given in Tables 5.1 and 5.2. The only fertiliser that was applied during the 1987/88 year was "Hyphos S" which contains no K.

8.4.1.1 *Analysis of herbage potassium in excreta affected and nonaffected pasture*

Herbage samples were collected from a grazed dairy pasture on the No. 4 dairy farm and their K concentration measured as detailed below. After analysis of the herbage the variation in herbage K concentrations that occur within a paddock due to the uneven distribution of animal excreta was also determined.

The site chosen for the above survey was a paddock that had been sown down in new pasture in the autumn of 1985, and the samples were collected in October 1985. A recently-established, N deficient pasture was chosen; the yellow colour of the sward facilitated identification of urine patches (rich in N and dark green).

At the time of sampling the paddock was ready to be grazed. Prior to grazing 10 sites were selected at random throughout the paddock. At each site one pair of excreta affected areas of pasture were identified as being of a similar sward density and height. Sampling areas were defined using a 0.156 m^2 quadrat. A similar pair of quadrats were also identified in an area not affected by excreta. One quadrat of each pair was cut to ground level with hand shears and the other clearly marked with a peg. After grazing the herbage on the remaining quadrat was cut to ground level.

Each herbage sample was mixed and a proportion removed for chemical analysis. This subsample was dried at 65^{o} C and then shaken over a 1 mm sieve to remove the adhering dirt. The K content was determined by the method described in section 3.2.4.

8.4.2 **Development of the Model For the Soil-Plant-Animal System**

8.4.2.1 *Estimation of potassium ingested by the dairy cows*

The mean milkfat production was 150 kg cow^{-1} or 450 kg milkfat ha^{-1}. Therefore the amount of pasture that was ingested by the cows was 11,250 kg DM ha^{-1}, based on 25 kg DM per kg milkfat (Bryant, 1982). While the DM ha^{-1} ingested by the cows is comparatively simple to estimate, the K concentration of this herbage is more complex. The K content of the pasture varies not only throughout the different seasons of the year (section 2.6.1.2) but also spatially, due to the pattern of dung and urine deposition throughout the paddock. For the purposes of developing a transfer model the area under consideration has been divided into two regions, i.e., pasture that has excreta deposited on it and the pasture that remains unaffected by excreta throughout the year.

The data from Chapter 3 indicates that over the period of maximum pasture growth the herbage is able to recover the K that remains in the top 15 cm of soil from a urination event. A similar result has been assumed for dung based on the information in the literature on the recovery of K by plants growing in dung affected soil (section 2.6.4.2). If the mean duration of the effect of excreta on herbage K uptake is less than one year then, although the deposition of new excreta continues, provided the stocking rate and number of excretions remains constant, the number of excretions which affect herbage K in one year remains constant and equal to the number deposited within the year (Petersen et al., 1956b).

The proportion of the area that is covered by excreta (P) can be calculated from an estimate of the density of excreta depositions (D) using the formulae of Petersen et al. (1956a) and Richards and Wolton (1976; $P=100-100/[(D+k)/k]^{k}$, where $D=N^{*}a/A$ and k=7) which have been defined in section 2.6.3.5.

From a knowledge of the management practices of the farm it was possible to calculate the number of cows that grazed one hectare in the year. For the No. 4 dairy farm the annual (1987/88) stock density was calculated at 1094 cows ha^{-1} yr^{-1} (Table 8.14). The total number of excretions per year, "N", was calculated from the data in Table 2.3 which shows that on

Table 8.14 Calculation of annual stock population density for the No.4 dairy farm (cows ha^{-1}).

	Rotation length (days) 12	24	45	90	
No. of rotations	10	4	1.3	1	
Months of year	Aug-Nov	Dec-Mar	Mar-Apr	May-Jul	
Stock density each rotation	36	72	135	270	
Total stock density	360	+ 288	+ 176	+ 270	= 1094

Table 8.15 The concentration of K in herbage cut to ground level before and after grazing from both excreta affected and non-affected pasture (%K).

		Before grazing	After grazing	$t_{0.05}$
Excreta affected pasture		3.20	2.50	NS
	SEM	0.18	0.20	
Non-affected pasture		1.80	1.60	NS
	SEM	0.12	0.09	

average each cow urinated 9.9 times per day and defaecated 13.1 times per day. The area covered by each excretion ("*a*") was based on 0.16 m^2 per urination (Figure 7.7) and 0.07 m^2 per defaecation (Table 2.5). Using the above formulae the density of excreta deposition per hectare (*D*) was calculated to be 0.27 m^2 m^{-2}, and the proportion of the pasture that was covered in excreta in one year (*P*) was 23%.

The area covered by excreta makes up 23% of the total area but was expected to grow 50% of the total DM production (i.e., 5,600 kg DM ha^{-1}) because the majority of K and N was recycled onto this area (Saunders, 1984). The mean K content of this pasture was estimated at 2.3%, which was the mean herbage K content for urine affected pasture measured over the growing season following a urination event (Chapter 3; Figure 3.3).

The remainder of the K ingested by the dairy cow was derived from the 77% of the area that was not affected by excreta. The K concentration of this herbage has been estimated at 1.8% which was a mean value of the herbage collected from non-urine affected areas in the field (Table 8.15).

From the above calculations it was estimated that 230 kg K ha^{-1} was ingested by the cows (130 kg K ha^{-1} from excreta affected pasture and 100 kg K ha^{-1} from nonaffected pasture; Figure 8.2). The amount of K ingested is used to calculate the animal K losses using the K Loss model (Appendix VIIa). The calculated values are detailed in Appendix VIId.

8.4.2.2 *Loss of potassium due to animal products*

The amount of K lost through animal products (milk, cull cows and calves) was estimated at 4.23 kg K cow^{-1} (Equations 1,2,3, Appendix VIIa) or 13 kg K ha^{-1} (Figure 8.2). This loss was derived from mean production values of 150 kg milkfat cow^{-1}, a 4.8% milkfat content, a cow liveweight of 440 kg and a lifespan of 5 milking seasons per cow.

8.4.2.3 *Loss of potassium due to transfer of excreta to unproductive areas*

Over the whole milking season (260 days) the cows spent an average of 3.5 hours per day in the shed and raceways. This amounted to spending 10% of the total year off the paddock and resulted in an estimated loss of 7.8 kg K cow^{-1} (Equation 4, Appendix VIIa). In addition the cows spent 25 days off the pasture on a feedpad, which resulted in a further loss of 5.1 kg K cow^{-1} (Equation 4, Appendix VIIa). The total K lost due to deposition of excreta in the dairy shed, raceways and feedpad amounted to 12.9 kg K cow^{-1} or 39 kg K ha^{-1} (Figure 8.2).

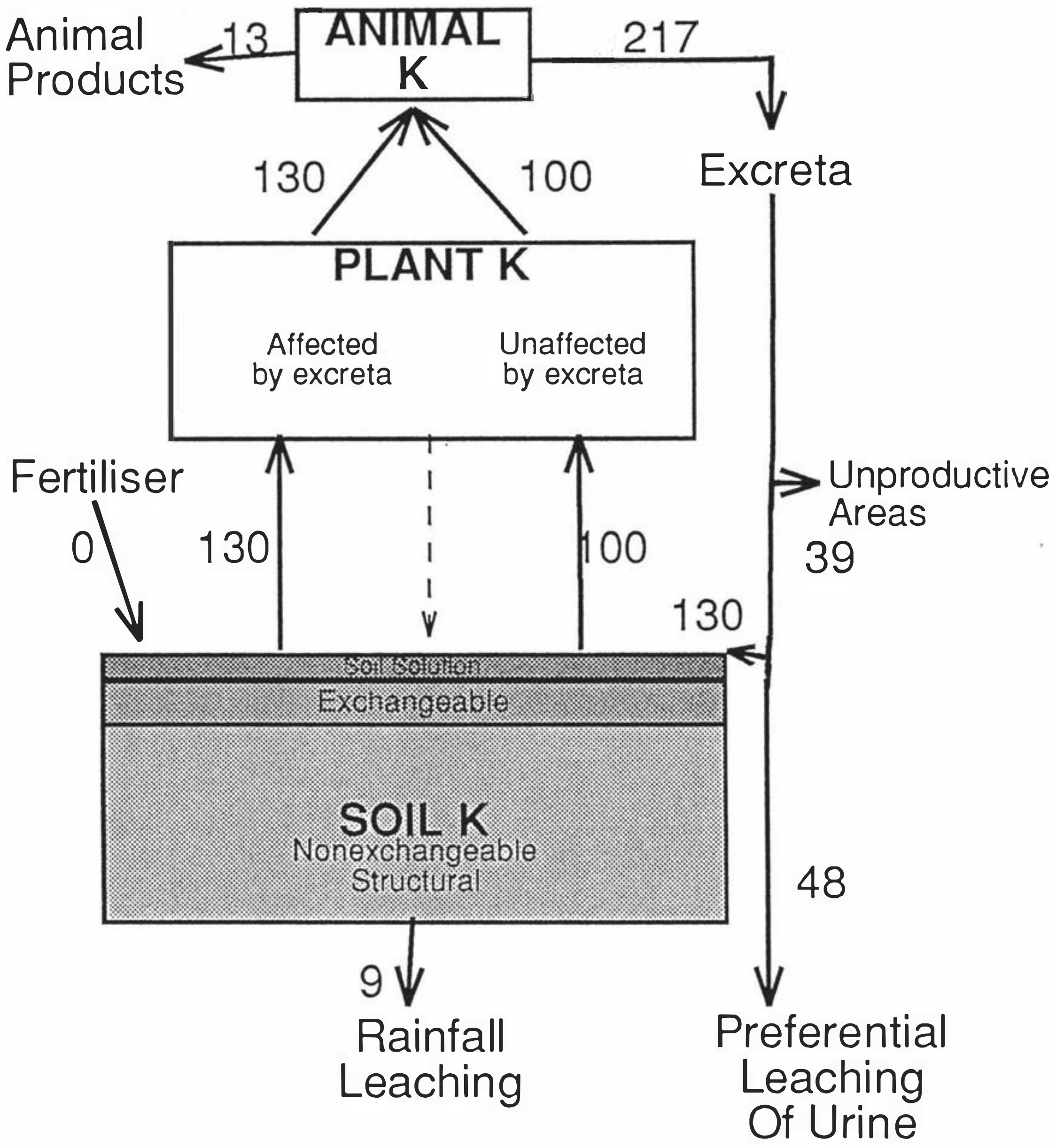


Figure 8.2 A model of net transfers of potassium on the Massey University No. 4 dairy farm (kg K ha^{-1}).

An analysis of the effluent from the dairy shed shows that there was an animal loss of 6 kg K cow^{-1} yr^{-1} (Clarke and Warburton, 1982). The total amount of K lost in the dairy shed and raceways was 7.8 kg K cow^{-1} yr^{-1}, therefore it seems that most of the excreta is deposited in the dairy shed rather than in the raceways.

8.4.2.4 *Loss of potassium due to preferential flow of urine and leaching from rainfall*

The amount of K deposited on the paddock in excreta was estimated at 178 kg K ha^{-1} (Figure 8.2) of which 90% (160 kg K ha^{-1}) was in the form of urine (Hutton et al., 1967). The proportion of this urine K that was lost from the effective plant root zone by preferential flow was estimated at 30% (Figure 7.2) or 48 kg K ha^{-1} (Figure 8.2).

The remainder of the urine K (112 kg K ha^{-1}) was retained in the top 15 cm of soil and probably would be recovered by the herbage within one year after application (Chapter 3; section 3.3.3.2). All of the K deposited in dung (18 kg K ha^{-1}) was considered to have been recovered by the herbage (MacDiarmid and Watkin, 1972a; During et al., 1973; Weeda, 1977;).

It was estimated that 9 kg K ha^{-1} were leached per annum from an annual drainage of 300 mm with a mean K concentration of 3 $\mu g\ ml^{-1}$ (Smith et al., 1984). As explained in section 8.2.2.4 there may be some error in calculating separate leaching losses for preferential flow and rainfall.

8.4.3 **Discussion**

The transfer model in Figure 8.2 is not complete. There are other components which have not been included such as the proportion of plant K that is not ingested by the animals. Since these components do not directly influence the K losses from the system they have not been considered.

It is obvious from Figure 8.2 that the dairy cow was the greatest cause of K losses (92% of total losses) from the farm, and that only a minor loss (8%) was due to leaching from rainfall. In fact, of the total K ingested by the cow, 43% was lost off the farm (Figure 8.2). The main loss mechanism for the ingested K was preferential flow of urine to soil depths below the effective plant rooting depth. For both the Taranaki Agricultural Research Station and the Wallace farm the major loss of K was also through this mechanism. These results indicate that preferential flow is a major loss pathway.

A greater proportion of the K losses occurred through the transfer of excretal K to unproductive areas of the farm on the No. 4 dairy farm compared with the Taranaki Agricultural Research Station and the Wallace farm. The main reasons for this were the longer milking time and the use of a feedpad on the No. 4 dairy farm. Thus a greater potential exists on the No 4 dairy farm for reducing the amount of K that is transferred to unproductive areas of the farm than at the other two sites.

The total amount of K lost annually from the No. 4 dairy farm was 109 kg K ha^{-1} Figure 8.2. Annual dressings of fertiliser K have not been applied to the farm, yet small plot observation and mowing trials carried out at several sites on the No. 4 dairy farm in the last 5 years have not provided any evidence that the farm is responsive to K fertiliser (P.E.H. Gregg, pers. comm.). It must be assumed therefore, that the rate of release of soil K from the 0-15 cm depth of soil is sufficient to meet a loss of 109 kg K ha^{-1} yr^{-1}. The amount of soil K present in the Tokomaru silt loam as exchangeable K^+ is approximately 300 kg K ha^{-1} (SEM=42) to a depth of 15 cm. If the K supply to the pasture plants is derived only from exchangeable K^+, then the supply of exchangeable K^+ if it was not replaced from nonexchangeable sources would last a maximum of 3 years at an annual loss of 109 kg K ha^{-1}. Analysis of the amount of exchangeable K^+ present in the 0-7.5 cm depth of soil over the last 5 years has shown little indication that these levels have declined (P.E.H. Gregg, pers. comm.). Therefore, adequate K to meet the 109 kg K annual loss must be released from the nonexchangeable sources of K^+ in this soil. The Tokomaru silt loam contains K rich clay minerals (illite; Table 5.1) which result in a large soil K content (17,000 kg K in the 0-15 cm depth of soil; Metson, 1980), the majority of which is present as nonexchangeable K^+. This suggests that the No. 4 dairy farm is unlikely to become K deficient in the short term, in the absence of fertiliser, provided the rate of supply of nonexchangeable K^+ continues to replace the losses of K. It is not known how long these K losses will continue to be replaced by K from the clay minerals.

The CFAS K model predicts that 17 kg K ha^{-1} should be applied as fertiliser per annum to the No. 4 dairy farm (Appendix VIII). The CFAS model calculated that the amounts of K lost from the No. 4 dairy farm were 84 kg K ha^{-1} yr^{-1} due to animal production and transfer to unproductive areas, and 3 kg K ha^{-1} yr^{-1} due to leaching in drainage water (Appendix VIII). When these losses are compared with those predicted by the K Loss model in Figure 8.2, the data suggest that the CFAS model may underestimate the total amount of K lost due to the animal (84 kg K ha^{-1} yr^{-1} for the CFAS model compared with 100 kg K ha^{-1} yr^{-1} for the K Loss model). The CFAS model also predicted that a lower amount of K was supplied by the soil (70 kg K ha^{-1} yr^{-1}; Appendix VIII) than the amount predicted by the model developed in this study (109 kg K ha^{-1} yr^{-1}; Figure 8.2).

8.5 GENERAL DISCUSSION

Through the construction of mass balance models of K losses and gains and a model of K transfers between the compartments of the K cycle in a grazed dairy pasture, it has been possible to estimate the major losses of K out of the K cycle and identify the factors which affect these losses. This information can be used to suggest ways of reducing these losses in order to decrease the amount of fertiliser required and also to give more precise fertiliser recommendations.

8.5.1 Reducing Potassium Losses on Dairy Farms

The estimations of K losses in this study have highlighted the role of the dairy cow in causing losses of K from grazed pasture. On the 3 farms for which estimates of losses have been made the dairy cows appear to be responsible for 85-91% of the total annual K loss on the Taranaki Agricultural Research Station (Table 8.7), 75% of the total loss over 30 years for the Wallace farm (Table 8.11) and 92% of the annual loss on the No. 4 dairy farm (Figure 8.2). The major loss occurs because dairy cows have a very small requirement for K in their metabolism, so the majority of the K they ingest is excreted in dung and urine. Potassium in the latter is subject to losses caused by both preferential flow of urine through the soil and transfer of urine and dung to unproductive areas.

8.5.1.1 *Loss of urine by preferential flow through soil macropores*

Of the losses created by dairy cows the greatest loss (approximately 50%) occurs through the preferential flow of urine K to soil depths beyond the effective plant rooting zone (Tables 8.7, 8.11; Figure 8.2). There are few if any practical ways of reducing this loss. The factors affecting preferential flow of urine K are related mainly to the ability of the urine to infiltrate through the soil surface into the soil (section 7.3.3). While the extent of preferential flow may be reduced in some situations through compaction of the soil surface and a water repellent soil surface, any reduction in K loss may be more than offset by reduced pasture growth caused by an excessive soil moisture content or a deficient soil moisture content under these respective conditions. Several methods have been suggested by Quin (1982) for increasing the ground area that is covered by urine such as breeding animals that walk while urinating and attaching devices to the animals that deflect the urine flow. These have, however, attracted little attention from the dairy industry to date.

8.5.1.2 *Transfer of excreta to unproductive areas*

The second major loss relating to the animal is the amount of K that is transferred to unproductive areas such as the raceways, dairy shed and feedpad. This loss of K could be reduced by minimising the time that the cows spend in these areas. For example, using the data for the Wallace farm, for every 30 minutes the cows spend in the dairy shed and raceways per day over the milking season there is a loss of 2 kg K cow^{-1} yr^{-1} (equivalent to \$1 cow^{-1} yr^{-1}). Similarly, for every day on a feedpad there is a loss of 0.3 kg K cow^{-1} yr^{-1}. Suggestions that could be carried out to ensure that transfer losses are kept to a minimum include an efficient layout of raceways and gateways leading to and from the shed, minimising the time the cows spend standing in the yards before and after milking, and better drainage of land to reduce the need for a feedpad. The benefits of drainage in reducing the transfer of excretal K to feedpads for a dairy farm on the Tokomaru silt loam have been calculated by Williams et al. (1988). They showed that during a wetter than average year, drainage decreased the time that cows spent on a feedpad by 49 days during the year. For a farm with a stocking rate of 3 cows ha^{-1}, this would reduce the amount of K transferred to the feedpad by 33 kg K ha^{-1} yr^{-1}. The collection and return of effluent from the dairy shed and feedpad to a large proportion of the farm will also help to reduce losses. While some of these suggestions may not be practical or economic for farmers to implement, it is important that they realise the significance of the losses that can occur through transfer to unproductive areas of the farm.

8.5.1.3 *Reducing the amount of potassium in excreta*

The quantity of K lost via dairy cow excreta is a function of the amount of K ingested, therefore the more K ingested the greater are the losses of K. It may be possible to reduce the K level in the ingested herbage by reducing K fertiliser inputs or using split fertiliser applications in situations where herbage K concentration is above the critical level for pasture growth. If the mean K concentration of the pasture on the Wallace farm is reduced from 3.2% to 2.8% (the latter being the critical value for pastures well supplied with N such as on most high producing dairy pastures; G.S. Smith et al., 1985), there is a 12% reduction in the K intake per cow (Table 8.16). While this reduced K intake has little effect on the amount of K lost in animal products, it would reduce the amount of K transferred to unproductive areas by 1 kg K cow^{-1} yr^{-1} and the amount of K lost by preferential flow of urine by 2 kg K cow^{-1} yr^{-1},

Table 8.16 Effect of reducing the herbage K concentration on K intake and K losses for the Wallace farm (kg K cow^{-1} yr^{-1}).

	Mean %K in herbage	
	3.2	2.5
K Ingested	113	88
K losses		
animal products	4	4
transfer	7	5
preferential urine loss	12	9

giving a total saving of 3 kg K cow^{-1} yr^{-1} (Table 8.16). A reduction in losses of this magnitude may result in significant savings in fertiliser costs (equivalent to \$1.50 cow^{-1} yr^{-1}) on farms with high stocking rates and/or large herd sizes.

An alternative method of reducing herbage K concentration is to develop pasture plant varieties which have a lower demand for K. Reducing herbage K may also have the benefit of reducing metabolic problems that can occur due to a high K concentration in relation to the Ca, Mg and Na concentrations in the herbage (section 2.6.1.3).

Lower amounts of K in urine may also reduce the amounts of Ca^{++} that are displaced from the soil surface exchange sites and leached from soil where urine is deposited. The data collected from the leaching experiments with intact soil cores in controlled-climate chambers (Chapters 5, 6) showed that between 43 and 371 mg Ca were leached following an application of 200 ml urine to the Tokomaru silt loam (Table 6.6). Extrapolating this to the field where urine events are 2000 ml not 200 ml suggests that between 430 and 3710 mg Ca may be leached per annum from one urine event. The number of urine events that occur in a year on the No. 4 dairy farm can be estimated using the stock density of 1094 cow days ha^{-1} yr^{-1} and the fact that a cow urinates 9.9 per day (section 8.4.2.1). Therefore between 5 and 41 kg Ca ha^{-1} may be leached per annum depending on the rate of drainage (Chapter 6). Magnesium can also be leached from urine spots and a calculation based on the data in Figure 5.6 shows that 4 kg Mg ha^{-1} yr^{-1} may be leached. The amount of Mg leached is calculated from the volume and Mg concentration of the leachate collected during a period of only one month and so may underestimate the amount leached over a longer period. While the amounts of Mg and Ca leached from urine patches are relatively small compared with the amount of native exchangeable Ca^{++} and Mg^{++} found in the soil types used in the intact core experiments (Table 5.5), on dairy farms where the soil Mg and Ca contents are low (e.g., peat soils) these losses may be more significant.

8.5.2 Comparisons of Estimated Net Potassium Losses with Changes in Soil Potassium Content

While it is possible to estimate the losses and gains of K from a grazed dairy pasture, it is difficult to verify the net loss or gain with actual measurements of the changes in soil K. The reasons for this are as follows:

Firstly, measurements of the amount of soil K, especially exchangeable K^{+}, are extremely variable because of the uneven spatial return of excretal K which makes detection of short term changes very difficult.

Secondly, inputs of K fertiliser over the last 50 years of dairy farming in New Zealand soils are negligible compared with the total amount of native soil K in most mineral soils (Table 2.1).

Thirdly, there is often a lack of suitable information on the original K status of the site being considered.

On both the Taranaki Agricultural Research Station and the Wallace farm an attempt was made to compare the estimates of the K losses and gains with measurements of change in soil K (sections 8.2.2.5 and 8.3.2.7). Both of these attempts illustrate the difficulty in comparing the estimated net losses or gains with changes in soil K. On the peat soil, however, with its negligible native K content, it should be possible to detect net changes in soil K. There were difficulties in accurately calculating the losses and gains of K for the Wallace farm. The calculated losses of K were based on incomplete data on both animal production, prior to 1974, and herbage K concentration. A further difficulty occurred in obtaining an estimate of the original K content of the peat soil. Despite these difficulties the total amount of K in the soil measured in the soil after 30 years of dairy farming was <13% of the total amount of K added as fertiliser over this time period. Improving the estimate of the change in soil K will not significantly affect the predictions made by the mass balance model that little of the fertiliser K had accumulated in the soil.

Short term monitoring of the net changes in soil K is possible on peat soils. Therefore a peat soil could be used to test the predictions of the K Loss model further by comparing the predicted change in soil K with the measured. This method would be particularly useful for determining the fertiliser K requirements for dairying on peat soils. In addition on a peat soil it would be possible to carry out field experiments that are designed to investigate ways of reducing K losses from dairy systems.

8.5.3 Improvements to the CFAS Model

The information obtained from the experiments and the mass balance models developed in this study can provide suggestions for the improvement of the CFAS model. Such improvements should increase the accuracy of K fertiliser recommendations. The suggested improvements relate to the amount of K lost via the animals, the measurement of the amount of K in the ingested herbage and an improvement in the quantification of the contribution of soil K to the cycle.

8.5.3.1 *Losses of potassium due to animals*

The information collected in this study indicates that the amount of K lost via animal products and transfer of excretal K to unproductive areas (the ALF) is overestimated by the CFAS model. For example, on both the Taranaki Agricultural Research Station and the Wallace farm the K Loss ALF was calculated at 1.6 kg K su^{-1} (section 8.2.2.3; Table 8.13), which was considerably lower than the CFAS ALF value of 4 kg K su^{-1}. The reason for the difference between the two ALFs is that the CFAS ALF was calculated using an 18% loss of excretal K due to transfer to unproductive areas of the farm, while the K Loss ALF was estimated from a transfer loss of 5-6% on these two farms. As discussed in section 8.2.2.3, the 18% in the CFAS model is based on estimates derived from New Zealand and overseas literature rather than direct measurements of the time that cows spend in unproductive areas on New Zealand farms.

The CFAS model takes little account of the loss of urine K due to preferential flow through the soil. This present study has shown that preferential flow is a major pathway for K losses. For seven commercial dairy farms in the Waikato, Taranaki and Manawatu region on a wide range of soil types, the calculated amount of K that was lost per year due to preferential flow of urine K beyond the effective plant rooting depth was between 12 and 49 kg K cow^{-1} yr^{-1} (Appendix IX). Data obtained in Chapter 6 indicate that the amount of K that is lost on a farm due to preferential flow of urine will depend more on the stocking rate than soil type, therefore a preferential flow loss factor could be incorporated readily into the CFAS ALF.

The CFAS model uses a constant value for the ALF of 4 kg K su^{-1} for dairy farms on flat topography and 4.5 kg K su^{-1} for dairy farms on hill country. However, for a specific farm the actual loss may vary according to the length of time that cows are off the pasture during milking and/or the use of a feedpad. For the seven commercial farms mentioned above, there was a range of milking times and number of days that a feedpad was used. This variation resulted in K losses due to transfer to unproductive areas of between 7 and 14 kg K cow^{-1} yr^{-1} (Appendix IX). This variation in transfer loss indicates that it may be desirable to have a variable ALF that can be calculated for individual farms. The ALF for a farm can be calculated simply from information provided by the farmers on animal production and the length of time that cows spend in the raceways, dairy shed and on a feedpad. The seven farmers mentioned previously supplied the information required readily.

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8.5.3.2 *Herbage potassium*

Quantifying the amount of K ingested by a dairy cow is dependent on both the amount of dry matter consumed and its K content. Whereas the dry matter consumed can be determined easily, the value used for the herbage K concentration can not (section 8.4.2.1). Herbage K concentrations vary both temporally and spatially (sections 2.6.1.2, 2.6.1.3). For the Taranaki Agricultural Research Station and the Wallace farm the mean annual K concentrations in the herbage were measured at 3.42% and estimated at 3.2%, respectively. These values are higher than the 3% used in the CFAS model (Campkin, 1985) and lower than the herbage K concentrations (1.8-3.2%) measured on the No. 4 dairy farm (Table 8.15). This range of measured K concentrations reflects the large range of herbage K concentrations that exist between farms (sections 2.3.2, 2.6.1.3). For an individual farm it is possible to model the spatial variability in herbage K concentration due to excreta deposition as shown in section 8.4.2.1 for the No. 4 dairy farm. The variation in herbage K between farms should be taken into account when predicting fertiliser requirements for individual farms.

Most information on herbage K concentration is derived from plant samples taken to ground level. As shown in Table 8.15, the K concentration of the whole sward down to ground level tended to be higher than the K concentration in the pasture left behind after grazing (i.e., the pasture between grazing height and ground level). Therefore using a K concentration determined from the whole sward down to ground level may underestimate the amount of K ingested. When measuring the amount of K ingested by cows only the grazeable herbage (i.e., the herbage above the 2-3 cm grazing height) should be sampled and analysed for K content. The higher K concentration of the grazeable herbage is due to the K being found mainly in young growth tissue rather than in stalk and dead material, which is found at the base of the sward and is not ingested by grazing dairy cows (McNaught, 1970). The difference between the K concentration of pre-grazed and residual pasture (both measured to ground level) was greatest in pasture that was affected by excreta (Table 8.15), which is the major source of the annually ingested herbage K (Figure 8.2). The variation in the data shown in Table 8.15 was large (CV=18-25%) and so more replicates (>10) would have been necessary to determine the statistical relevance of this difference in K concentration between pre-grazed and residual pasture. A significant difference in nutrient concentration between utilised and residual pasture has been reported by Rowarth (1987) for P in hill country sheep pastures and this affects the quantity of P ingested and recycled by animals.

8.5.3.3 *Contribution of soil potassium to the cycling potassium pool*

The mass balance models in sections 8.2 and 8.4 showed a net annual loss of up to 50 kg K ha^{-1} yr^{-1} from the Taranaki Agricultural Research Station farmlet and 109 kg K ha^{-1} yr^{-1} from the No. 4 dairy farm. At these sites there was no evidence to suggest that the soil exchangeable K^+ pool (as measured by the QTK) decreased in size or, for the No. 4 dairy farm, that the site was responsive to K fertiliser. Therefore it appeared that soil nonexchangeable K^+ was releasing approximately 50 and 109 kg K ha^{-1} yr^{-1} on the Taranaki Agricultural Research Station and the No. 4 dairy farm respectively. Using the information provided by the CFAS K model on these soil types it was estimated that the release of soil K to the cycling pool would be 105 and 70 kg K ha^{-1} yr^{-1}, respectively. The differences between the mass balance model and CFAS estimates of the nonexchangeable K^+ released at each site will result in major differences in the K fertiliser recommendations. For example, on the Taranaki Agricultural Research Station the fertiliser recommendation varies by 55 kg K ha^{-1} depending on which K supply value (50 or 105 kg K ha^{-1} yr^{-1}) is used. Further errors in calculating the soil K supply in the CFAS model occur due to the use of QTK values in the equation (section 2.8). The QTK has an expected error of 30% (section 2.6.1.4) therefore the calculated amount of K supplied by the soil can vary from 64 to 136 kg K ha^{-1} yr^{-1} for the Taranaki Agricultural Research Station.

The difference between the results from this study and the amount of K supplied by the soil as calculated by the CFAS model illustrates the need for an increased understanding of the magnitude and rate of contribution of nonexchangeable K^+ to the K cycle in a grazed pasture. The rate of release of K from different clay minerals needs to be quantified for New Zealand soils under grazing Factors which may affect this rate of release such as type of clay mineral, climatic conditions and the amount of K in exchangeable form should be included in such a study. The use of ^{40}K, which has recently become commercially available, may provide a suitable technique because the ^{40}K could be used to label the K pool of soils containing different K minerals. The relative release and plant availability of K could then be measured.

8.6 CONCLUSIONS

The amount of K lost from three grazed dairy farms was calculated using the K Loss model developed in this study of the fate of K in grazed dairy pasture. The total estimated annual losses of K were in the order of 75-109 kg K ha^{-1} yr^{-1}. Between 75 and 92% of these losses were caused directly by the dairy cow through product removal from the farm, excreta transfer to unproductive areas and preferential flow of urine K through the soil to beyond the effective plant rooting depth.

On the Taranaki Agricultural Research Station and the Wallace farm comparisons were made between the estimated K losses from the farm with actual measurements of changes in soil K. These comparisons were difficult due to the high spatial variation of exchangeable K in a grazed dairy pasture, and the high K content of New Zealand mineral soils against which the amounts of K added in fertiliser or estimated as lost were small. However, on the Wallace farm the losses of K as estimated by the K Loss model gave a good prediction of the change in soil K after 30 years of receiving fertiliser. The peat soil with its low K content is the soil type with the greatest potential for observing changes in soil K caused by fertiliser addition and animal systems. It is suggested that field experiments on peat soils could be used to investigate the various factors affecting K losses from dairy farms.

The total amount of K that was lost via the animal, as calculated by the K Loss model, differed from the losses predicted by the CFAS model for two of the three farms in this study. The CFAS model predicted greater animal K losses on the Taranaki Agricultural Research Station, but lower animal K losses on the No. 4 dairy farm. As well there were major differences in the relative contribution from the three components which make up the animal K loss parameter (animal products, excreta transfer to unproductive areas and preferential flow of urine K through soil). The CFAS model appeared to overestimate the amount of K transferred to unproductive areas, but did not directly account for the preferential loss of urine K, although the latter accounted for approximately 50% of the K losses on the three farms in this study. The K Loss model has the advantage in that the three components of the animal loss parameter are derived and calculated separately rather than collectively as in the CFAS model. This separation allows identification of the relative causes of K losses from the farm and the opportunity to consider ways of reducing these losses. In the K Loss model, the calculation of the animal loss components requires information that can be supplied easily by the farmer and so could be adopted readily by advisers for providing fertiliser recommendations.

CHAPTER 9

SUMMARY

The information obtained from this study has been summarised in two parts. The first part describes the fate of K in a dairy pasture once it is returned to the soil in the form of urine. The second part deals with the "K Loss" model that was developed to predict losses of K that occur when K is transferred and redistributed in a grazed dairy system by the action of soil, plant and animals. Comparisons between the K Loss model and the CFAS model, used by the MAF to predict fertiliser requirements, are also summarised.

FATE OF URINE POTASSIUM

The information in the literature on the fate of K cycling in dairy systems is extremely variable due to the wide range of conditions under which the data were collected (such as soil type, climate and type of experiment). For example, the majority of the leaching studies have been carried out in the laboratory or glasshouse (many using repacked columns of soil) and extrapolation of these results to the field is not always realistic. There is also a lack of information on the depth of the soil profile from which pasture plant roots extract the majority of their K requirements. This is an important factor in determining the extent to which, and the frequency with which, excretal K is recycled.

Initially two experiments were carried out to determine the depth of soil from which plants extract the majority of their K requirements and if potassium chloride could be used to simulate dairy cow urine as a source of K.

Plant uptake of potassium

A field experiment showed that pasture plant roots growing in urine affected Tokomaru silt loam soil absorbed the majority of their K from within the 0-15 cm depth of soil. Even under dry summer conditions, when the moisture content of the topsoil was low, very little K was extracted from beyond 15 cm. Therefore in soils with pasture rooting patterns similar to the Tokomaru silt loam any K that moves beyond the 15 cm depth of soil due to leaching can be considered to be lost from the cycle.

Effect of potassium source on adsorption of potassium by soil

Within 24 hours after application of urine to a laboratory incubation study, the soil pH increased by one pH unit as the urea in the urine hydrolysed to NH_4^+. On a soil which contained a large amount of pH dependent negative surface charge (Egmont brown loam, a yellow-brown loam) this increase in soil pH was accompanied by an increase in cation exchange capacity and K^+ adsorption. When KCl was incubated with the same soil the pH change was not so large and so smaller amounts of K^+ were adsorbed by the soil. On a soil type with smaller amounts of pH dependent charge (Tokomaru silt loam, a yellow-grey earth), no difference in K^+ adsorption was found between the two sources of K. Such a finding suggested that in the field, the extent to which urine K is adsorbed by soils and thus lost by leaching may vary between soil types according to soil chemical properties. Therefore actual urine and not potassium chloride should be used to measure leaching losses of K. Furthermore, experiments should include a range of soils with contrasting chemical properties, particularly those having differences in their amounts of pH dependent negative surface charge.

Preferential Flow of Urine

Immediately after a urination event a variable proportion (0-80%) of the urine flowed via a network of soil macropores through intact cores (15 cm in diameter and depth) of four contrasting soil types. The elution patterns of K^+ and N (mainly in the form of urea) were similar to those of Cl^- and the tritiated water ($^3H_20$) which was used as a tracer to monitor urine movement. It was concluded that this preferential flow of urine occurred too rapidly for there to be significant adsorption reactions between the soil surface and the K^+ and N in the urine. This implied that if 20% of the urine moved preferentially beyond the 15 cm depth of soil, then 20% of the K^+, N, Cl^- and other solutes (e.g., sulphur) in the urine would also move beyond this depth.

Measurements of the amount of urine K that moved beyond the effective plant root zone by preferential flow in the field were carried out on seven different soil types. Each of these soil types had contrasting soil physical and chemical properties. On four of the soil types the effect of season on the amount of preferential loss was also investigated. Results showed that even when urine was allowed to flow freely over the soil surface, between 0 and 46% of the urine was lost preferentially beyond the 15 cm depth of soil. A strong linear relationship ($r = 0.80^{***}$) was obtained between the surface area of the soil covered by a urination event and the proportion of the urine that was retained within the 0-15 cm depth of soil. This relationship suggested that the amount of urine that moved preferentially through the soil profile was dependent on the surface conditions of the soil (e.g., compacted or water repellent soil surface, or uneven surface microtopography) rather than the soil type or the moisture content of the soil. Thus where the conditions of the soil surface prevented infiltration (e.g., pugging or water repellency) the urine covered a larger area of the soil surface and the loss of urine by preferential flow was small. It may be possible to use this relationship for rapid estimations of the amount of urine K lost by preferential flow at a particular site by measuring the surface area covered by a simulated urination.

Leaching Losses From Rainfall

Of the urine that remained within the 0-15 cm depth of soil after the urination event, the majority could be found in the soil in the form of exchangeable K^+. The amount of K that was leached beyond the 15 cm depth of urine affected soil by subsequent rainfall events was determined in two experiments using intact soil cores of four soil types with contrasting chemical and physical soil properties (yellow-grey earth, recent, yellow-brown loam and yellow-brown sand soils). The results showed that the amount of K leached from simulated urine events was small (up to 361 mg K $core^{-1}$) compared with the amount leached preferentially following a urination event (up to 977 mg K $core^{-1}$). The amount of K leached was dependent on the rate at which the simulated rain water moved through the soil cores rather than by the type of soil involved. Where the rain water moved rapidly through the soil's macropores, the leaching losses from rain events tended to be small (3-6% of the applied urine K on an annual basis) as the rain water bypassed the urine K which had interacted with the soil surfaces. In situations where less of the rain water moved by preferential macropore flow, the water came into contact with the K that had moved into the soil micropores and so the amount leached was higher (equivalent to 7-27% of applied urine K on an annual basis). The latter conditions could occur where drainage is impeded or rainfall intensity is low.

Other cations were also leached from urine affected soil in rainfall induced drainage, notably NH_4^+ originating from the urine application, and Ca^{++} which was displaced from soil surface cation exchange sites by K^+ and NH_4^+ ions. The major anions leached were NO_3^-, Cl^- and HCO_3^- or OH^-.

Plant Uptake of Urine Derived Potassium

The urine K retained in the 0-15 cm depth of soil as exchangeable K^+ (41% of urine K applied) from a July application of urine, was recovered by the plant roots during the following spring and summer. Such a result suggests that the plant roots will be able to recover all the urine K from the soil surface exchange sites within a year after application.

PREDICTING LOSSES OF POTASSIUM FROM A GRAZED DAIRY PASTURE

The information obtained on the fate of urine K in this study was used to construct a K Loss model which predicted the amount of K lost annually from grazed dairy systems via animal products, transfer of excretal K to unproductive areas of the farm and preferential loss of urine. Net loss from dairy systems leads to reductions in soil K. Thus in theory, K loss over many years will lead to measurable decreases in soil K. On most soil types in New Zealand, however, changes in the quantity of K present in the 0-15 cm depth of soil over time are difficult to measure. This is due not only to the high spatial variation in soil exchangeable K^+ in grazed dairy pasture but also because of the continuous replenishment of exchangeable K^+ from the extremely large pool of nonexchangeable and structural forms of soil K. The latter problem is illustrated by an example using a yellow-grey earth on the Massey University No. 4 dairy farm (Tokomaru silt loam) in which the background total K content of the soil is 17,000 kg K ha^{-1} in the 0-15 cm depth of soil. In comparison with this figure the calculated annual loss of 109 kg K ha^{-1} yr^{-1} is extremely small and difficult to measure.

One of the few soil types suitable for comparing predicted losses and actual changes in soil K is an organic soil, as this soil has a very low native K content (approximately 300 kg K ha^{-1}) and a virtual absence of soil minerals that release nonexchangeable K^+. A mass balance of K losses and gains for a farm on this soil type showed good agreement between the losses predicted by the K Loss model and the measured changes in soil K.

Comparison with CFAS Model

A comparison of the parameters used in the K Loss model and the CFAS model to predict K losses on three seasonal supply dairy farms revealed a number of differences. The CFAS model predicted that losses of K due to transfer of excretal K to unproductive areas were much higher than in this study (e.g., the CFAS model predictions are based on 3.2 kg K su^{-1} yr^{-1} compared with 1.9 kg K su^{-1} yr^{-1} which was calculated by the K Loss model for the Massey University No. 4 dairy farm). The reason for the difference between values is that in the CFAS model the assumption is made that dairy cows return 18% of their excreta to unproductive areas, whereas on the farms used in this study the cows returned less than 10% of their excreta to unproductive areas. For the above example the CFAS model may overestimate the transfer loss by 1.3 kg K su^{-1} yr^{-1} or 39 kg K ha^{-1} yr^{-1}.

Whereas the CFAS model may overestimate the amount of K lost through transfer, the model underestimates the amount of K lost via leaching by not allowing for the loss that occurs during preferential flow of urine through the soil profile. The present study has shown this to be a major mechanism for K losses accounting for 50% of the K losses from the cycle (equivalent to 29-48 kg K ha^{-1} yr^{-1} for the three farms studied).

The CFAS model uses a constant value for the amount of K lost via animal products and transfer to unproductive areas (i.e., 4 kg K su^{-1} for all dairy farms on flat topography and 4.5 kg K su^{-1} for dairy farms on hill country). Data collected from dairy farms shows that the amount of K lost per animal via animal products (milk, calves and cull cows) is fairly constant between farms (3-4 kg K cow^{-1} or 0.5-0.7 kg K su^{-1}). However, the amount of K lost due to transfer to unproductive areas is variable (7-14 kg K cow^{-1} or 0.9-1.9 kg K su^{-1}) according to management practices (e.g., the time taken to milk the cows and the use of a feedpad during wet periods of the year). This variation indicates that the animal losses of K should be calculated independently of production loss for each individual farm. The animal induced losses for a farm can be calculated simply from information that is readily available from dairy farmers (e.g., animal production data and the length of time that cows spend off the paddock in the dairy shed etc.). The amount of K lost due to preferential flow of urine to beyond the effective plant rooting depth could also be included in this calculation of animal losses.

Losses of K are inevitable in a grazed dairy pasture due mainly to the redistribution of K by the grazing animal. The data from this study can be used to predict the amount of K that will be lost for a particular farm incorporating aspects of productivity and management that are not currently considered by the CFAS model. From these losses it is possible to calculate the amount of K fertiliser that is required to maintain dairy production at the desired level.

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Appendix I The total amount of leachate collected during the three day period following urine deposition on the surface of intact soil cores as described in Chapter 5.

Treatment	Soil type	Leachate per core (g)
Water	Tokomaru	177.9
	Manawatu	148.3
	Egmont	155.7
	Pukepuke	164.2
Low urine	Tokomaru	164.8
	Manawatu	150.8
	Egmont	164.6
	Pukepuke	141.5
High urine	Tokomaru	184.4
	Manawatu	145.6
	Egmont	171.2
	Pukepuke	144.3
		SED 10.16 CV 2%
Soil main effect	Tokomaru	175.7
	Manawatu	148.2
	Egmont	163.8
	Pukepuke	150.0
		SED 5.87

Appendix II The chemical composition of leachates collected from the growth chamber experiment described in Chapter 5. (Data not reported in Chapter 5.)

a) **Ammonium** (mg N core $^{-1}$)

Cumulative rainfall	Soil	Treatment			CV	SED	Significance	
		water	low	high	%		urine	soil
0 mm	Tokomaru	1.7	7.1	63.9	37	45.01	*	***
	Pukepuke	1.3	44.2	33.9				
	Egmont	0.3	77.1	43.7				
	Manawatu	0.3	94.9	222.0				
10 mm	Tokomaru	0.43	2.42	5.74	17	1.73	***	***
	Pukepuke	0.66	9.84	9.92				
	Egmont	0.26	1.95	3.88				
	Manawatu	0.35	2.75	5.71				
20 mm	Tokomaru	0.29	4.04	8.39	8	1.46	***	***
	Pukepuke	0.38	7.41	12.96				
	Egmont	0.28	2.51	5.08				
	Manawatu	0.23	2.20	5.56				
30 mm	Tokomaru	0.27	3.83	8.61	8	1.19	***	***
	Pukepuke	0.29	6.05	13.16				
	Egmont	0.16	1.93	4.28				
	Manawatu	0.13	1.85	4.69				
40 mm	Tokomaru	0.03	2.50	7.59	8	1.24	**	***
	Pukepuke	0.12	3.14	9.65				
	Egmont	0.11	3.10	5.74				
	Manawatu	0.10	1.12	4.08				
50 mm	Tokomaru	0.10	3.53	8.80	5	1.22	***	***
	Pukepuke	0.17	4.22	9.68				
	Egmont	0.39	2.10	5.28				
	Manawatu	0.09	0.93	3.82				
60 mm	Tokomaru	0.02	2.96	7.67	13	1.35	***	***
	Pukepuke	0.03	4.67	9.24				
	Egmont	0.44	2.04	5.21				
	Manawatu	0.10	0.66	4.20				
70 mm	Tokomaru	0.05	1.86	6.91	15	0.95	***	***
	Pukepuke	0.05	2.89	8.64				
	Egmont	0.05	1.95	6.65				
	Manawatu	0.08	0.51	3.43				
80 mm	Tokomaru	0.04	1.69	8.23	19	1.37	***	***
	Pukepuke	0.05	2.09	10.78				
	Egmont	0.09	2.20	8.32				
	Manawatu	0.05	0.47	3.31				
90 mm	Tokomaru	0.05	1.30	6.45	24	1.34	*	***
	Pukepuke	0.02	1.19	8.20				
	Egmont	0.04	1.19	7.52				
	Manawatu	0.04	0.19	2.88				

b) **Calcium** (mg Ca core^{-1})

Cumulative rainfall	Soil	Treatment			CV	SED	Significance	
		water	low	high	%		urine	soil
0 mm	Tokomaru	1.64	1.02	3.52	17	2.16	***	***
	Pukepuke	0.84	1.59	2.72				
	Egmont	1.05	12.59	8.21				
	Manawatu	2.08	17.41	19.66				
10 mm	Tokomaru	1.67	1.93	2.65	38	1.28	*	NS
	Pukepuke	0.44	0.83	1.81				
	Egmont	0.54	0.69	0.80				
	Manawatu	1.27	2.86	4.90				
20 mm	Tokomaru	2.52	3.85	5.07	23	1.17	***	**
	Pukepuke	0.40	1.43	1.18				
	Egmont	0.45	2.01	1.68				
	Manawatu	1.05	2.21	4.16				
30 mm	Tokomaru	1.33	5.48	8.70	25	1.61	***	***
	Pukepuke	0.34	0.81	1.26				
	Egmont	0.42	2.51	2.38				
	Manawatu	1.01	4.63	6.12				
40 mm	Tokomaru	0.79	5.62	9.73	29	1.90	***	***
	Pukepuke	0.19	0.53	1.23				
	Egmont	0.49	2.34	3.12				
	Manawatu	0.45	2.93	4.27				
50 mm	Tokomaru	1.48	9.50	12.26	11	1.20	***	***
	Pukepuke	0.32	1.64	2.99				
	Egmont	0.39	3.28	4.13				
	Manawatu	0.35	3.08	4.75				
60 mm	Tokomaru	0.41	5.03	8.63	26	1.50	***	***
	Pukepuke	0.13	1.54	3.72				
	Egmont	0.35	3.03	4.77				
	Manawatu	0.74	5.69	8.78				
70 mm	Tokomaru	0.54	6.06	11.23	27	1.63	***	***
	Pukepuke	0.11	1.13	3.20				
	Egmont	0.27	3.31	4.93				
	Manawatu	0.51	4.63	8.72				
80 mm	Tokomaru	0.42	7.32	13.60	28	2.39	***	***
	Pukepuke	0.15	1.43	5.28				
	Egmont	0.40	6.90	7.43				
	Manawatu	0.37	8.05	13.79				
90 mm	Tokomaru	0.34	8.55	12.24	32	1.92	***	***
	Pukepuke	0.07	1.01	3.71				
	Egmont	0.10	1.22	2.67				
	Manawatu	0.18	1.85	4.98				

c) **Magnesium** (mg Mg core^{-1})

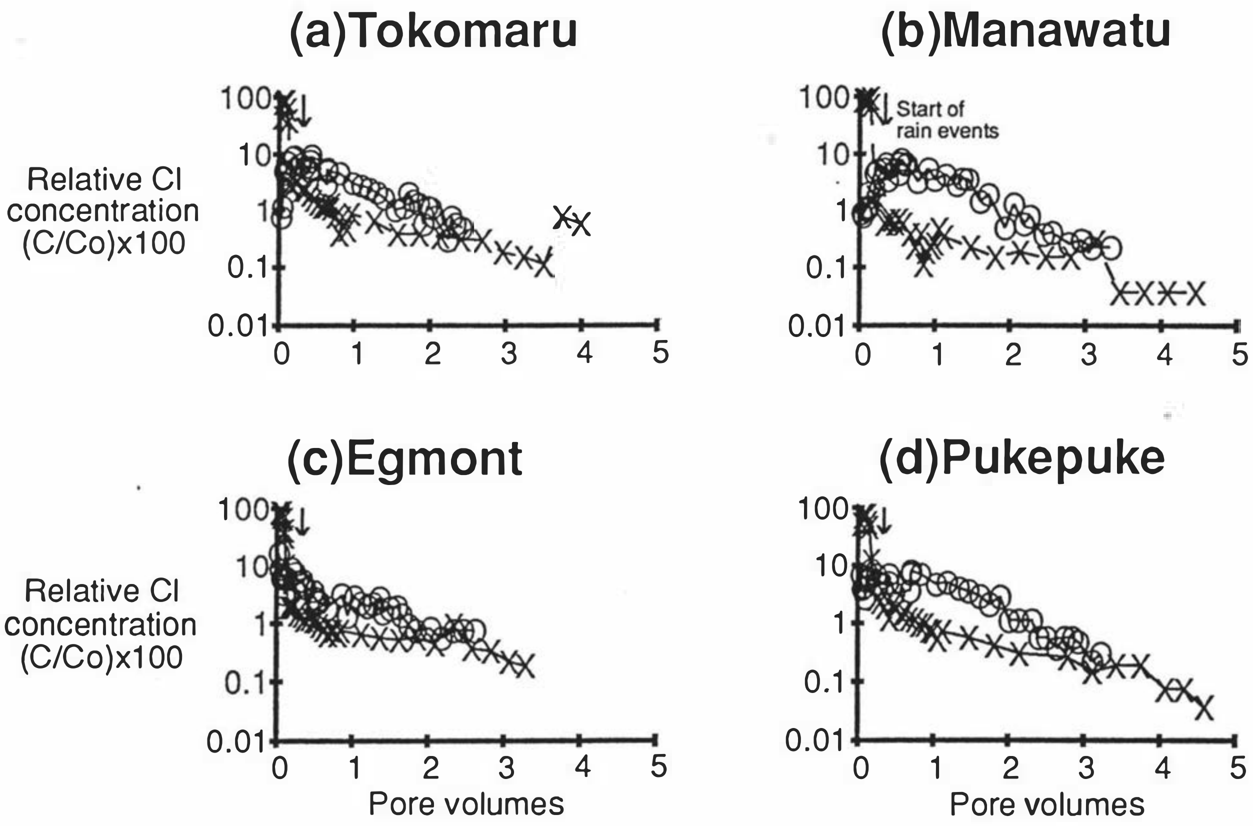
Cumulative	Soil	Treatment			CV	SED	Significance	
rainfall		water	low	high	%		urine	soil
0 mm	Tokomaru	0.33	7.60	12.62	28	2.66	***	***
	Pukepuke	0.15	7.50	6.60				
	Egmont	0.07	3.95	3.24				
	Manawatu	0.26	10.43	19.90				
10 mm	Tokomaru	0.38	0.39	0.62	33	0.22	*	*
	Pukepuke	0.04	0.58	0.61				
	Egmont	0.06	0.10	0.12				
	Manawatu	0.19	0.35	0.67				
20 mm	Tokomaru	0.41	0.78	1.10	30	0.26	**	***
	Pukepuke	0.07	0.68	0.72				
	Egmont	0.08	0.30	0.36				
	Manawatu	0.24	0.53	1.06				
30 mm	Tokomaru	0.28	0.97	1.36	33	0.30	***	***
	Pukepuke	0.05	0.47	0.70				
	Egmont	0.08	0.31	0.50				
	Manawatu	0.25	1.15	1.30				
40 mm	Tokomaru	0.18	1.10	1.73	33	0.39	*	***
	Pukepuke	0.05	0.40	0.44				
	Egmont	0.11	0.39	0.85				
	Manawatu	0.14	1.09	1.02				
50 mm	Tokomaru	0.25	1.41	2.09	23	0.28	***	***
	Pukepuke	0.04	0.46	0.48				
	Egmont	0.12	0.72	0.80				
	Manawatu	0.14	1.88	2.34				
60 mm	Tokomaru	0.11	0.86	1.46	20	0.34	***	***
	Pukepuke	0.03	0.16	0.73				
	Egmont	0.08	0.76	1.16				
	Manawatu	0.22	2.10	3.21				
70 mm	Tokomaru	0.11	0.90	2.07	19	0.36	***	***
	Pukepuke	0.02	0.44	0.80				
	Egmont	0.07	0.92	1.20				
	Manawatu	0.18	1.73	3.27				
80 mm	Tokomaru	0.09	0.90	2.24	20	0.43	***	***
	Pukepuke	0.01	0.43	1.02				
	Egmont	0.09	1.21	1.23				
	Manawatu	0.12	1.56	2.95				
90 mm	Tokomaru	0.06	1.15	2.40	17	0.40	***	***
	Pukepuke	0.01	0.33	0.95				
	Egmont	0.03	0.40	0.82				
	Manawatu	0.06	0.76	2.21				

d) **Chloride** (mg Cl core^{-1})

Cumulative rainfall	Soil	Treatment			CV %	SED	Significance	
		water	low	high			urine	soil
0 mm	Tokomaru	2.7	15.1	48.32	26	8.81	*	***
	Pukepuke	2.6	20.3	67.4				
	Egmont	1.2	17.7	28.4				
	Manawatu	1.0	14.3	28.7				
10 mm	Tokomaru	1.7	7.0	23.6	18	5.25	**	***
	Pukepuke	1.5	20.5	38.2				
	Egmont	0.8	6.9	12.4				
	Manawatu	0.7	12.0	22.0				
20 mm	Tokomaru	1.6	24.0	27.5	15	5.34	*	***
	Pukepuke	2.0	13.1	32.5				
	Egmont	0.6	8.4	18.0				
	Manawatu	0.6	8.4	22.5				
30 mm	Tokomaru	1.0	10.6	25.9	20	4.91	**	***
	Pukepuke	1.6	17.1	37.1				
	Egmont	0.4	7.3	15.3				
	Manawatu	0.5	9.0	20.2				
40 mm	Tokomaru	0.5	7.5	24.4	19	5.80	NS	***
	Pukepuke	0.8	11.2	19.3				
	Egmont	0.5	7.6	25.2				
	Manawatu	0.4	5.4	19.0				
50 mm	Tokomaru	0.7	19.5	17.0	24	4.83	NS	***
	Pukepuke	1.1	11.3	20.3				
	Egmont	0.7	9.9	29.6				
	Manawatu	0.3	6.7	18.9				
60 mm	Tokomaru	0.7	5.8	18.1	32	5.29	NS	***
	Pukepuke	0.9	5.5	20.1				
	Egmont	0.7	8.4	29.1				
	Manawatu	0.4	6.8	22.4				
70 mm	Tokomaru	0.3	5.4	18.6	20	4.36	NS	***
	Pukepuke	0.3	5.1	26.3				
	Egmont	0.4	7.4	30.0				
	Manawatu	0.4	5.1	18.1				
80 mm	Tokomaru	0.4	7.9	18.1	36	5.20	*	***
	Pukepuke	0.6	4.7	28.4				
	Egmont	0.4	15.0	33.0				
	Manawatu	0.2	4.3	17.6				
90 mm	Tokomaru	0.2	4.3	15.4	46	5.37	NS	***
	Pukepuke	1.5	3.1	24.9				
	Egmont	0.2	10.4	21.5				
	Manawatu	0.2	2.3	12.5				

Appendix III The concentration (%) of Ca, Mg and Na in herbage harvested from the cores at the end of the experiment described in Chapter 5.

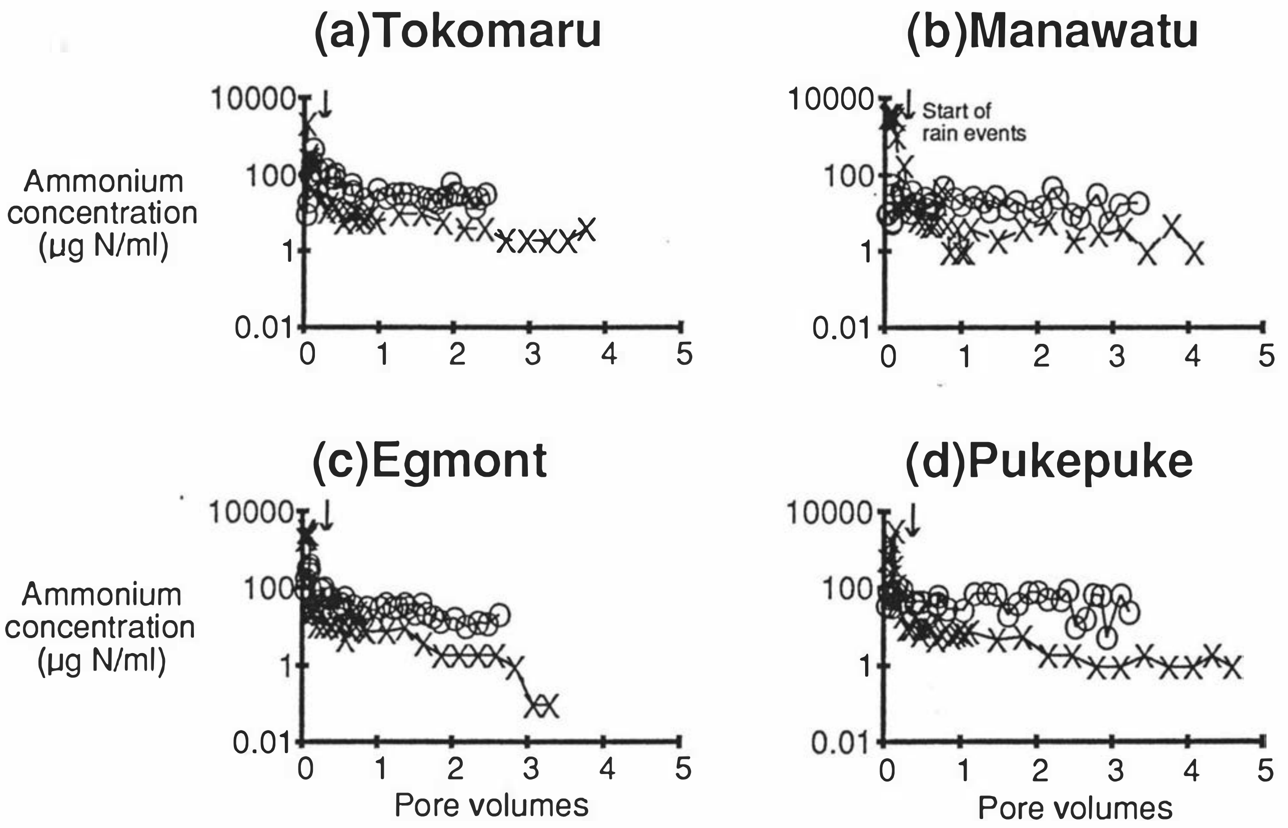
Soil	Treatment	Calcium		Magnesium		Sodium	
		clover	grass	clover	grass	clover	grass
Tokomaru	water	1.22	0.66	0.20	0.15	0.22	0.23
	low urine	1.30	1.14	0.24	0.18	0.16	0.12
	high urine	1.17	0.55	0.24	0.10	0.10	0.08
Manawatu	water	0.72	0.27	0.22	0.18	0.11	0.15
	low urine	0.95	0.44	0.34	0.18	0.11	0.07
	high urine	0.75	0.62	0.28	0.23	0.13	0.13
Egmont	water	0.95	0.32	0.19	0.16	0.15	0.07
	low urine	0.65	0.41	0.18	0.17	0.05	0.03
	high urine	0.49	0.38	0.15	0.11	0.07	0.06
Pukepuke	water	1.12	0.66	0.16	0.14	0.17	0.15
	low urine	0.92	0.44	0.20	0.11	0.22	0.08
	high urine	0.66	0.45	0.17	0.12	0.10	0.12
SED		0.122	0.129	0.037	0.038	0.044	0.090
Significance of main effect:							
soil		***	***	***	*	*	NS
urine		***	NS	*	NS	*	NS



Appendix IVa Breakthrough curves of chloride for fast draining (X) and slow draining (O) cores after the application of 200ml of dairy cow urine and sufficient simulated rainfall to produce approximately 300ml of drainage (equivalent to 2.4-4.6 pore volumes) as described in Chapter 6.

Appendix IVb The amounts of Cl leached from soil cores immediately following the urine application and during the simulated rain events (expressed as mg Cl per core and % of Cl applied).

Soil name Core no. (Drainage rate)	mg Cl in drainage Urine application	 Rain events	 Total	% of applied Cl Urine application	 Rain events
Tokomaru					
6 (fast)	248.9	161.6	410.5	50	32
3 (slow)	9.9	243.8	253.7	2	49
10 (slow)	15.2	260.3	275.5	3	52
14 (slow)	46.3	235.3	281.6	9	47
Manawatu					
2 (fast)	213.1	60.5	273.6	43	12
7 (fast)	290.6	55.6	346.2	58	11
16 (fast)	398.8	91.9	490.7	80	18
12 (slow)	3.31	244.2	247.5	1	49
Egmont					
4 (fast)	177.3	197.6	374.9	35	40
9 (fast)	309.7	98.1	407.8	62	20
5 (slow)	17.2	314.8	332.0	3	63
15 (slow)	29.5	271.3	300.8	6	54
Pukepuke					
1 (fast)	271.2	94.5	365.7	54	19
8 (fast)	347.4	67.4	414.8	69	13
11 (slow)	12.9	305.6	318.5	3	61
13 (slow)	166.5	202.2	368.7	33	40



Appendix V

The concentration of NH_4^+-N in leachate from fast draining (X) and slow draining (O) cores after the application of 200ml of dairy cow urine and sufficient simulated rainfall to produce approximately 300ml of drainage (equivalent to 2.4-4.6 pore volumes) as described in Chapter 6.

Appendix VI The retention of bromide in the 0-15 cm depth of several pastoral soils.

Site	% Br retained	Mean	SEM	%CV
Winter				
Tokomaru	59, 68, 70, 73, 75	69.0	2.77	9
Manawatu	46, 55, 59, 79, 96	67.0	9.04	30
Egmont	91, 92, 93, 99, 100	95.0	1.87	4
Pukepuke	58, 58, 62, 64, 70	62.4	2.23	8
Waihou	90, 94, 98, 99, 100	96.2	1.87	4
Taupo	39, 42, 42, 46, 50, 54, 55, 58, 64, 72	52.2	3.34	20
Rukuhia	71, 72, 78, 84, 85, 86, 95, 97, 98, 100	86.6	4.77	12
Summer				
Tokomaru	60, 61, 67, 78, 89	71.0	5.35	17
Manawatu	77, 81, 90, 101, 107	91.2	5.71	14
Egmont	62, 78, 79, 83, 96	79.6	5.45	15
Pukepuke	94, 104, 106, 111, 113	105.6	3.33	7

Appendix VIIa Glossary of terms and equations used in the K Loss model to calculate kg K lost cow^{-1} yr^{-1} as described in Chapter 8.

Terms

M	kg K in milk cow^{-1} yr^{-1}
MF	kg milkfat produced cow^{-1} yr^{-1}
mf	mean milkfat test for the year (%)
C	kg K in culled cows and calf production cow^{-1} yr^{-1}
W	mean weight of mature cow (kg)
A	number of seasons the cow is milked
c	number of calves produced by a cow in her lifetime
cw	mean calf weight at birth (kg)
P	kg K lost in animal products cow^{-1} yr^{-1}
T	kg K transferred to unproductive areas of the farm cow^{-1} yr^{-1}
h	K concentration in the herbage (% w/w)
t_1	mean time (hours) that the first cow to be milked spends in the dairy shed and raceways at each milking
t_2	mean time (hours) that the last cow to be milked spends in the dairy shed and raceways at each milking
d	length of milking season (days)
f	number of days the cows spend on a feedpad yr^{-1}
PL	kg K lost due to preferential flow of urine beyond the 15 cm depth of soil cow^{-1} yr^{-1}
plf	proportion of urine that moves by preferential flow beyond the 15 cm depth of soil
TAL	total kg K lost cow^{-1} yr^{-1}

Equations

1. $M = (MF \times 0.0013 \times 100)/mf$

[0.0013 represents the K content of milk]

2. $C = [(W/A) \times 0.0015]+[(c \times (A-1)/A) \times cw \times 0.0017]$

[0.0015 and 0.0017 represent the K contents of cow and calf body tissue]

3. $P = M + C$

4. $T = [(MF \times 25 \times h/100)-P] \times [[[(t_1+t_2)/24] \times d/365.25]+ f/365.25]$

[25 represents the conversion of dry matter consumed to kg milkfat produced; 24 and 365.25 represent the number of hours per day and days per year]

5. $PL = [[[(Mf \times 25 \times h/100)-P]-T] \times 0.9] \times plf$

[0.9 represents the proportion of excretal K that is in the form of urine]

6. $TAL = P + T + PL$

Appendix VIIb Output of K Loss model for the Taranaki Agricultural Research Station as described in Chapter 8. Values denoted by "*" were not required as better information was able to calculate the K losses.

```
CALCULATION OF K LOST PER COW PER YEAR  Taranaki
                                        1985/86
PRODUCTION/COW                          kg K/cow  equation
** kg fat per cow ?                              *
** mean fat test (%) ?                           *
  kg milk produced                        3876.00
  kg K in milk                               5.04         1

** weight of mature cow ?                       4
** no. of years in herd ?                       5
  kg cow culled                               0.8
  kg K in cow culled                       0.0012         2

  no. of calves produced                        3
  kg K in calf production                    0.03         2

Total production K/cow                       5.07         3

EXCRETA TRANSFER LOSS/COW
** conversion of DM to kg milkfat ?
   (suggestion is 25)                            *
  total K ingested                            150
  total excretal K                         144.93
time cows spend off the paddock during each milking
** 1st cow back to paddock (hours) ?         0.75
** last cow back to paddock (hours) ?        0.75
  mean time off paddock per day               1.5
  proportion of day off paddock              0.06
** no. of days cows are milked per yr ?       273
  proportion of year off paddock             0.05
** no. of days cows spend on a feedpad per year ?
                                                0
  K deposited on feedpad                     0.00

K deposited in unproductive areas/cow        6.78         4

PREFERENTIAL FLOW OF URINE LOSS
  excretal K returned to paddock           138.16
  urine K returned to paddock              124.34
** proportion leached below 15cm ?           0.12

  K leached preferentially                  14.92         5

TOTAL K LOST PER COW PER YEAR               26.76         6
```

Appendix VIIc Output of K Loss model for the Wallace farm as described in Chapter 8.

```
CALCULATION OF K LOST PER COW PER YEAR  JD & RD Wallace
                                        1986/87

PRODUCTION/COW                          kg K/cow     equation
** kg fat per cow ?                          141
** mean fat test (%) ?                       4.5
  kg milk produced                       3133.33
  kg K in milk                              4.07        1

** weight of mature cow ?                    400
** no. of years in herd ?                      4
  kg cow culled                              100
  kg K in cow culled                        0.15        2

  no. of calves produced                       3
  kg K in calf production                   0.03        2

Total production K/cow                      4.26        3

EXCRETA TRANSFER LOSS/COW
** conversion of DM to kg milkfat ?
   (suggestion is 25)                         25
  total K ingested                         112.8
  total excretal K                        108.54
time cows spend off the paddock during each milking
** 1st cow back to paddock (hours) ?         0.5
** last cow back to paddock (hours) ?        1.5
  mean time off paddock per day                2
  proportion of day off paddock             0.08
** no. of days cows are milked per yr ?      270
  proportion of year off paddock            0.06
** no. of days cows spend on a feedpad per year ?
                                               0
  K deposited on feedpad                    0.00

K deposited in unproductive areas/cow       6.69        4

PREFERENTIAL FLOW OF URINE LOSS
  excretal K returned to paddock          101.85
  urine K returned to paddock              91.67
** proportion leached below 15cm ?          0.13

  K leached preferentially                 11.92        5

TOTAL K LOST PER COW PER YEAR              22.86        6
```

Appendix VIId Output of K Loss model for the Massey University No. 4 dairy farm as described in Chapter 8.

```
CALCULATION OF K LOST PER COW PER YEAR  No. 4 dairy

PRODUCTION/COW                                 kg K/cow  equation
** kg fat per cow ?                                 150
** mean fat test (%) ?                              4.8
  kg milk produced                              3125.00
  kg K in milk                                     4.06         1

** weight of mature cow ?                           440
** no. of years in herd ?                             5
  kg cow culled                                      88
  kg K in cow culled                              0.132         2

  no. of calves produced                              4
  kg K in calf production                          0.03         2

Total production K/cow                             4.23         3

EXCRETA TRANSFER LOSS/COW
** conversion of DM to kg milkfat ?
   (suggestion is 25)                                25
  total K ingested                                78.75
  total excretal K                                74.52
time cows spend off the paddock during each milking
** 1st cow back to paddock (hours) ?                  1
** last cow back to paddock (hours) ?               2.5
  mean time off paddock per day                     3.5
  proportion of day off paddock                    0.15
** no. of days cows are milked per yr ?             260
  proportion of year off paddock                   0.10
** no. of days cows spend on a feedpad per year ?
                                                     25
  K deposited on feedpad                           5.10

K deposited in unproductive areas/cow             12.85         4

PREFERENTIAL FLOW OF URINE LOSS
  excretal K returned to paddock                  61.68
  urine K returned to paddock                     55.51
** proportion leached below 15cm ?                  0.3

  K leached preferentially                        16.65         5

TOTAL K LOST PER COW PER YEAR                     33.73         6
```

Appendix VIII Calculation of fertiliser requirements for the Taranaki Agricultural Research Station and Massey University No. 4 dairy farm using the CFAS K model (details of which are given in section 2.8) as described in Chapter 8.

Parameter	Taranaki	No. 4
Animal losses		
SR	25.2	21
ALF	4	4
Leaching losses		
R	1260	1000
PE	800	800
Leaching factor	2	1
Soil supply		
QTK	12.5	7
Kc	2	0
Equation		
K (kg K ha^{-1}) = [SR x ALF] + [(R - PE) x ((SR / 1250) x Leaching factor)] - [10(QTK - Kc)]		
Fertiliser K		
kg K ha^{-1}	14	17

Appendix IX Estimated losses of K for commercial seasonal supply dairy farms using the K Loss model as described in Chapter 8 (kg K cow^{-1} yr^{-1}).

Location and soil type	Animal product loss	Transfer loss	Preferential loss of urine
Manawatu			
yellow-grey earth	4	13	17
yellow-brown sand	4	11	19
recent	3	8	14
Taranaki			
yellow-brown loam	4	10	18
Waikato			
yellow-brown loam	5	14	22
yellow-brown pumice	5	7	49
organic	4	7	12