DETERMINATION OF SUSTAINABLE NITROGEN LOADING RATES FOR LAND TREATMENT SYSTEMS WITHOUT ADEQUATE SOIL AND GROUND WATER INFORMATION: DAIRY FARM EFFLUENT APPLICATION ONTO GRAZED PASTURE IN THE WAIKATO REGION

N Selvarajah
Environment Waikato
P 0 Box 4010, Hamilton East

Introduction

In New Zealand where agricultural land resources are not scarce like many overseas nations, land treatment systems are the preferred method for treating effluent effectively. Land treatment systems are attractive for all parties involved in effluent management. Land treatment systems are culturally acceptable, provide additional income for effluent users, and meet regional councils' objectives in minimising point sources of ground water and surface water pollution. Land treatment systems are also of interest to scientists studying soil, plant and ground water reactions to applied effluent.

Although most New Zealand soil types are effective in treating effluent constituents such as bacteria, organic carbon and phosphorus, the treatment efficiency of applied nitrogen (N) appears to be limited under unsaturated conditions. Consequently, one of the major environmental concerns related to land treatment systems is ground water nitrate (N03-N) contamination due to excessive N loading. Elevated N03-N levels in ground water could restrict potable water supply and enrich N03-N in surface water. Moreover, since New Zealand's economy depends mainly on food production, we cannot afford to ignore the European Union's initiatives to manage N03-N from agricultural sources.

In New Zealand, little information is available on environmentally sustainable N loading rates for effluent application onto land. Despite intensive research on soil reaction to N fertilisers in New Zealand and world-wide, loading rates for fertiliser-N are yet to be determined for New Zealand soils. Soil reaction to effluent application is a more complex and dynamic process than that of fertiliser-N. Consequently, intensive research is required to determine suitable N loading rates for a wide range of effluent, different soil types and prevailing climatic conditions in New Zealand. Meanwhile, regional councils often use conventional systems (e.g. treatment ponds, constructed wetlands, activated sludge) to treat and discharge effluent into waterways that may not always bring the desired environmental outcome.

It is beyond the scope of this paper to discuss a wide range of effluent and their loading rates to a range of land treatment systems. This paper uses land application of dairy farm effluent[1] as an example to illustrate the concepts used to determine N loading rate for grazed dairy pasture system for resource management purposes. The paper discusses N inputs and outputs and provides a detailed N budget for a grazed clover-based dairy pasture system. The approach is based on the existing information in New Zealand on N transformations in soils. Originally this

[1] The paper was presented at the "Recent Developments in Understanding Chemical Movements in Soil" workshop at the Fertiliser Lime Research Centre, Massey University in February 1996.

paper also formed the technical basis for the determination of effluent-N loading rate for the Waikato region in the Changes to Transitional Regional Plan for the Dairy Farm Effluent Operative Plan. It is hoped that the paper also helps highlight the 'deficiency of soil-N transformation information in New Zealand.

Discussion

There is no scarcity for effluent in New Zealand. Effluent sources may be classified into three categories: farm (dairy, piggery, poultry and mushroom), industrial (dairy, meat, skin, hide, wool and food processing and rendering plants and stock and fertiliser truck washes) and domestic (sewage). Among the farm sources, dairy farm effluent contributes a significant proportion of nutrient, organic carbon and hydraulic loading to the total effluent stream in New Zealand. According to dairy statistics in New Zealand there are 14, 649 dairy farms with 2.83 million cows (Dairy Statistics, 1995). Approximately 42% of these farms are located in the Waikato region.

Terminology

It has been common practice among resource managers and the research agencies in New Zealand to use the following terms related to dairy farm effluent management: dairy shed effluent, dairy shed waste -water, dairy shed, cow shed and effluent disposal. From the dairy industry and sustainable resource management viewpoints these terms are inappropriate. Considering the international and national quality requirements imposed on milk, the high capital investment on dairy farming and the associated complex farm management practices, it is proposed that dairy shed or cow shed should be referred to as **milking parlour** or **milking area**. Consequently, the term dairy shed effluent should be referred to as **dairy farm effluent** or **farm dairy effluent**. The use of terms such as waste water should be discontinued because dairy farm effluent is a resource not 'waste'. The term effluent disposal is inappropriate if effluent is irrigated onto land to grow plants. However, such a term may be appropriate if treated effluent is discharged into waterways. It is suggested that when effluent is irrigated to produce drymatter, the activity must be referred to as **effluent irrigation** (or 'effluent application' and 'effluent spreading', or any other terms as long as the term disposal is avoided).

Nitrogen transformations and *annual* N budget for clover-based dairy pasture systems

Determining a N budget for any grazed pasture systems is a difficult task due to the interaction between pasture, climate, soil and grazing animals. Various attempts have been made in the past to develop a N flow model for the Manawatu (Field and Ball, 1982) and the U.K. (Ryden, 1984; Jarvis, 1993) grazed pastures. Jarvis (1993) summarised, "although there are data from experimental systems, a total comprehension of flows is not yet possible". There can also be a substantial variation in soil, pasture and climatic conditions and farm management practices.

This paper considers a typical Waikato farm as a model farm for estimating a N budget estimate. Stocking rates in the Waikato region can vary greatly, but are typically from 2.0 to 3.5 cows ha^{-1}. Milk production also varies from about 300-700 kg milk fat ha^{-1}. However, most dairy farms in the Waikato produce above average milk and have above average stocking rates. The following assumptions are made to determine N outputs from a grazed dairy pasture system for a model farm in me Waikato region: *Stocking rate is 2.7 cows ha*$^{-1}$*; well established clover-based dairy pasture system with well drained soil type; dry matter production is 13000 kg ha*$^{-1}$ *herbage N is*

4% of the dry matter; zero fertiliser N application; lactation period is 270 d; and milk is produced for factory supply.

Major pathways of N transformation in a grazed pasture are identified as: plant uptake, clover N fixation; N losses from excreta through ammonia (NH_3), volatilisation, leaching and denitrification; transfer of excretal N to unproductive areas (raceways and milking parlour) and removal of N through milk production.

Plant uptake

New Zealand dairy pasture systems comprise mainly a mixture of ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.). The extent of plant uptake of N is influenced by pasture growth rate which is influenced by soil moisture, temperature and extent of grazing. Generally peak pasture growth and greater uptake of N occurs during spring, whilst the lowest uptake of N occurs during winter. Pasture performance is measured in terms of dry matter (DM) production. In most cases, increases in DM production are apparent for N application rates up to 550-650 kg N ha^{-1} y^{-1} – beyond which the yield decreases (Ryden, 1984). At high N application rates pasture begins to accumulate N. Nitrogen accumulates in plants mainly in the form of protein. In addition to protein, NO_3-N can also accumulate in plats if there is excessive NO_3-N present in soils (Jarvis *et al*., 1989).

Compared with many other regions in New Zealand, the Waikato has optimal conditions for pasture growth. Many soils are derived from volcanic parent materials which when combined with typically warm, humid summers and mild winters ensure that Waikato soils are among the most productive in the country. Pasture uptake of N is approximately 520 kg N ha^{-1} y^{-1} in the Waikato region assuming that average dairy pasture production is 13000 kg DM ha^{-1} and a high N content of pasture of about 4% on a dry weight basis.

<u>N inputs</u>

Clover fixation of N

The ability of clover to fix atmospheric dinitrogen gas (N_2) in association with a symbiotic bacteria (*Rhizobium spp.*) in the form of organic-N is referred to as symbiotic N fixation. Clover stolons and roots decompose and contribute to the soil organic-N reserve. Mineralisation of decomposed clover plants, which is a slow N transformation process, is the key source of mineral-N in an ungrazed clover-pasture system. However, under grazed conditions clover-N is ingested by grazing animals and excreted as dung and urine. Consequently, in grazed pasture systems clover-N transfer to mineral-N forms is relatively rapid. The amount of N fixed per land area is influenced mainly by the size of the clover population and type of cultivar, soil moisture and temperature conditions, and the presence of soil mineral-N (Ledgard and Steele, 1992). Increased fertiliser-N can also reduce clover performance through reduction in clover-N fixation and growth (Ledgard, 1989; Ledgard *et al*., 1996).

The most widely used cultivar has been *Huia*, which has been used for a number of years. Recently, some new cultivars (eg *Pitau* and *Kopu*) have been introduced. Ledgard *et al*. (1990) showed that the amount of N fixed by *Aran, Kopu, Pitau* and resident clovers averaged 280 kg N ha^{-1} y^{-1} whilst that of *Huia* amounted to 224 kg N ha^{-1} y^{-1} when established on a Horotiu sandy loam. These values have been obtained from 2 y old cultivars and it is believed that when clover is well established it is capable of fixing a greater amount of N.

In grazed pasture, clover performance can also be affected by urine voided onto grass (Ball *et al*., 1979). However, it has been estimated that about 248 kg N ha^{-1} y^{-1} can be fixed by clover in an intensively grazed dairy pasture system in the Manawatu area (Field and Ball, 1982). More recently, Ledgard et al. (1996) using the ^{15}N dilution method estimated that without fertiliser-N input clover could fix up to 212 kg ha^{-1} y^{-1} in the Waikato area. Using this information, it is assumed that under zero N fertiliser use, on average, Waikato soils will be able to receive a minimum of 200 kg N ha^{-1} y^{-1} through clover-N fixation from an established clover-ryegrass pasture. It must be emphasised that since clover-N input is the major N input into grazed pasture systems in the absence of fertiliser-N input, careful consideration must be given in the estimation of clover-N input in other regions in New Zealand.

N Outputs

The major N outputs are N lost through milk production and excretion.

Nitrogen removal through milk production

In the Waikato region 'factory milk supply' is generally produced from about mid-August to mid-May. 'Town milk supply' is carried out throughout the year. Most dairy farms in the Waikato region cater for 'factory milk supply' and thus the average milking duration for the Waikato region is approximately 270 d. Farms that produce 'town milk supply' comprise a very small proportion of the total dairy farms in the region and hence are not considered in determining a N budget for dairy pasture.

Milk production per given area depends mainly on the breed of cow, stocking rate, and pasture production. In the Waikato region there are a large number of Holstein-Friesian herds which produce low fat low protein milk. The crossbred cows (e.g. Holstein-Friesian x Jersey) produce more milk fat and protein than the pure breeds (Dairy Statistics, 1995). On average, dairy farms in the Waikato region produce 425 kg milk fat ha^{-1} y^{-1} and 313 kg milk protein ha^{-1}. Assuming a N conversion of 0.16, the amount of N removed from the system is 50 kg N ha^{-1} y^{-1}. Approximately 20% of the milking herd are 2 y old cows, and unlike > 3 y old cows they use substantial amounts of N for body growth. However, 2 y old cows produce about 15 kg milk protein less than the 3 y olds per milking season. Thus it is estimated that approximately 2 kg N ha^{-1} is used for body maintenance. Assuming the calf body weight at birth is 35 kg $calf^{1}$, the N removal through 2.7 calves is approximately 5 kg ha^{-1}. The amount of N used for milk production, body maintenance and calving is estimated as 57 kg N ha^{-1} y^{-1} and rounded up to 60 kg N ha^{-1} y^{-1}.

Nitrogen losses from excreta

Dairy cows excrete a large proportion of N ingested. Jarvis *et al*. (1989) indicated that the amount of N consumed and excreted depended mainly on the N content of pasture; the higher the N content, the greater the excretion of N. According to their work about 60% of the N excreted is urine-N and 40% is dung-N for animals fed on a clover-ryegrass mixture (with zero N application). In ruminant animals, when N is ingested it is transformed into NH_3 and amino acids through microbial breakdown. Whilst amino acids are absorbed in the small intestine for protein synthesis in animals, NH_3 can be either converted into urea or recycled in the system. When intake of N exceeds that of the required level, the net N loss to NH_3 exceeds conservation of N in the system. The excess level of NH_3 produced in the rumen not only increases the urea output in urine, but triggers greater metabolisable energy use for recycle of NH_3 produced or synthesis of NH_3 into urea. Since clover contains a higher amino acid content than grass, a greater amount of N is absorbed in the digestive system and consequently plant protein losses as NH_3 are greater in animals fed with grass (Waghorn and Barry, 1987). The unabsorbed amino acids and plant protein (mainly clover protein) can escape through faeces (dung-N).

(a) Excreta-N transferred to unproductive areas (milking parlour and raceways)

Insufficient information is available on the amount of excreta-N that is transferred to non-pasture areas. Excreta can be deposited in substantial amounts in milking parlours, raceways and camping areas (e.g. around water troughs).

The amount of excreta deposited depends mainly on the length of time animals spend in these areas. For example it was estimated that the amount of time spent in the dairy and collecting yard is 2 h and hence about 8% of the excreta is deposited during milking time (Vanderholm, 1984). The length of time spent in the milking parlour varies greatly depending on the management practices. For example, data obtained from a Taranaki Regional Council (TRC) Technical Report (1990) showed that for a herd size of approximately 200 (n = 6) the time spent in the milking area was in the range of 1-3 h milking^{-1}. Assuming that the length of time spent in the milking area is directly related to the amount excreted, the range of excreta loss is estimated as 8-25% of total excretion per day. It has been assumed that the amount of excreta deposited in the milking area is the same in the morning and afternoon. However, more excreta is generally deposited in the morning than in the afternoon. It is also suspected that certain milking management practices can have a profound effect on animal behaviour which in turn can influence me amount of excreta generated during milking. No accurate estimates have been made on the amount of excreta transfer to raceways and watering areas. Depending on the herd management, animals can spend a substantial amount of time in raceways (e.g. transfer of herds for milking or grazing) and hence large amounts of excreta may be deposited in the raceways.

Nitrogen in dairy farm effluent

Soil mineral-N availability can indirectly influence the amount of N excreted. Increased rates of N fertiliser use can increase plant uptake of N and result in increased consumption of plant N. This can increase the amount of N excreted by grazing animals (Jarvis *et al*., 1989). However, mineral-N levels in soil increase during spring and summer and hence increase the plant absorption of N. Goold (1980) showed that N levels in dairy farm effluent fluctuated throughout the milking season following a bimodal pattern. Pasture composition can also regulate the amount of N excreted.

Another approach to estimating N deposited in the milking area is to consider the N content of the effluent produced during milking. The most widely used datum for dairy farm effluent N content is that of Vanderholm (1984), who showed that effluent contains 0.02% of N. Assuming 50 1 cow^{-1} d^{-1} of effluent is generated in the milking parlour, a dairy cow could excrete 10.4 g N d^{-1}. A one-off effluent survey performed by AgResearch, Ruakura in summer 1993 indicated that effluent contains 0.04% N and on average 20 g N cow^{-1} d^{-1} is excreted in the milking parlour

(Environment Waikato, 1994). According to this estimate the model farm will sustain a N loss of 14.6 kg ha^{-1} y^{-1}.

N loss in the milking parlour = stocking rate (cows ha^{-1}) x N excreted (kg N cow^{-1} d^{-1}) x lactation period (d)
= 2.7 cows ha^{-1} x 0.02 kg N cow^{-1} d^{-1} x 270 d y^{-1}
= 14.6 kg N ha^{-1} y^{-1}

If a similar amount is transferred onto raceways the total amount transferred to non-pasture areas will be approximately 30 kg N ha^{-1} y^{-1}. Field and Ball (1982) estimated that the transfer of N to non-pasture areas is 32 kg N ha^{-1} y^{-1}.

(b) Ammonia volatilisation from excreta deposited onto pasture

Amount of N excreted onto pasture per year	= N ingested (kg N ha^{-1}) - (N removed in milk (kg N ha^{-1}) + N excreted onto non-productive areas (kg N ha^{-1}))	
	= 520 - (60+30) kg N ha^{-1}	= 430 kg N ha^{-1}
Amount of urine-N excreted per year	= 430 x 60/100 kg N ha^{-1}	= 258 kg N ha^{-1}
Amount of dung-N excreted per year	= 430 x 40/100 kg N ha^{-1}	= 172 kg N ha^{-1}

Up to 85% of urine-N is in the form of urea (Doak, 1952). The balance of N in urine is in the form of allanton, hippuric acid, creatine, creatinine, amino-N and NH_3. Ammonia-N is <1% of the urine-N. Amino-N contributes significantly to the non-urea-N pool in urine (up to 15.9% of the urine-N). It has been assumed that of the 258 kg urine-N ha^{-1} about 85% hydrolyses into ammoniacal-N according to Eq. [1].

$$CO(NH_2)_2 + 2H_20 \xrightarrow{\text{urease}} NH_4^+ + NH_3\uparrow + HCO_3^- \qquad [1]$$

Volatilisation rates peak within 3 days of urination. Volatilisation can continue for 2-3 weeks at a slower rate. During rainfall or an irrigation event, the volatilisation process is arrested and urea-N is leached below the surface and it will continue to hydrolyse and produce ammoniacal-N. If the intensity of the rainfall or irrigation is very high, urea-N is leached below the rhizosphere. This occurs because urea is very mobile in soil.

Depending on the plant density, a proportion of the NH_3 volatilised from the soil surface can be absorbed by plants through stomatal openings. The greater the plant density (height and population per area), the higher the absorption (Hoult and McGarity, 1987). Urea-N deposited onto plant cover is also hydrolysed rapidly and the NH_3 produced is either absorbed by plants or volatilised to the atmosphere.

It becomes clear that the volatilisation process of urea-N voided during an urination event is complex. The extent of NH_3 loss is influenced by several factors. Urea-N concentration, micrometeorological conditions, presence of plants, soil type and soil moisture are the major factors. Field trials conducted on a wide range of New Zealand soils under controlled conditions clearly demonstrate that the amount of NH_3 loss from surface applied urea can vary greatly (Selvarajah, *et al.*, 1993). The same trials showed that the major soil characteristic that controls the amount of NH_3 volatilised from urea-N is the capacity for soils to resist an alkalinity build-up during the urea hydrolysis process, not the soil acidity as has generally been reported in the literature.

Sherlock and van Der Weerden (1992) showed that synthetic cattle urine applied at 500 kg N ha^{-1} rate onto pasture under Canterbury field conditions sustained 21.6% loss as NH_3-N (108 kg N ha^{-1}) within three days of application. Such a loss is comparable with that obtained with field application of urine under Manawatu conditions (Ball and Keeney, 1983).

It can be argued that the micrometeorological and soil conditions are conducive for high NH_3 volatilisation in the Canterbury and Manawatu regions compared to that of the Waikato region. A recent study in the Waikato region showed that NH_3 loss sustained from grazed pasture without fertiliser-N input was 15 kg ha^{-1} y^{-1}. This loss is considered to be very low and may not affect the N budget significantly. Nevertheless, from the agronomical viewpoint the model assumes a worst case scenario for the extent of NH_3 volatilisation in the Waikato region and considers 20% loss from urea-N.

Annual NH_3 volatilisation from urine voided onto pasture = (urea-N = urine-N x 85/100) x 20/100 kg N ha^{-1}
= (258 x 85/100) x 20/100 kg N ha^{-1}
≈ 44 kg N ha^{-1}

Ammonia is also volatilised from dung at much slower rates than that from urine, hence the relative potential for N loss from dung through the volatilisation process is small (Ryden *et al*., 1987). Approximately 3-8% of the dung-N is likely to volatilise under a range of Manawatu conditions (Sugimoto and Ball, 1989). This loss is only 1-3% of the total N excreted.

Annual NH_3 volatilisation from dung voided onto pasture = dung-N x 2/100 kg N ha^{-1}
= 172 x 2/100 kg N ha^{-1} ≈ 4 kg N ha^{-1}

∴ NH_3 lost from excreta voided onto pasture = NH_3 loss from urine and dung
= 44 + 4 kg N ha^{-1} y^{-1} ≈ 50 kg N ha^{-1} y^{-1}

Leaching losses from grazed dairy pasture

It has been estimated that urea-N is voided at the rate of 800 kg N ha^{-1} during each urination event (Doak, 1952). If 20% of this amount is volatilised, 640 kg N ha^{-1} will remain in the soil system. This is in addition to about 140 kg N ha^{-1} voided as non-urea-N in urine. Following urination due to the NN_3 build-up and the subsequent alkalisation of soil, organic carbon present in soil can be hydrolysed to more labile forms. This is favourable for biomass growth and consequently a large proportion of the ammoniacal-N produced can be immobilised by the microbes. ^{15}N trials performed in the field show that about 15% of the urea-N can be rapidly assimilated as ammonium and immobilised by microbes within 2 weeks of urea application onto the soil surface (Selvarajah, 1991). Microbial immobilisation of ammonium is a dynamic process and it continues until labile carbon (Okereke and Meints, 1985) or ammonium become deficient in the environment (Wickramasinghe *et al*., 1985). Ammonium released from urea-N can also be taken up by plants. Under unsaturated conditions most of the remaining ammonium is oxidised (nitrified) to form nitrate.

After urea hydrolysis, it takes about two weeks for nitrate (NO_3-N) to appear in the soil. Among the mineral-N species, NO_3-N has greater leaching potential than ammonium, because the ammonium ion (NH_4^+-N) is positively charged and hence adsorbed onto the cation exchange sites of clay and organic particles. Consequently, NO_3-N is very mobile in soil and can be leached below the rhizosphere during high rainfall or irrigation. The potential for soil N leaching increases when nitrate is generated in soil in excess of the amount required by plants. Often the conditions are not conducive for plant uptake of N in the urine patch because a high concentration of NH_3 retards plant growth and under dry conditions may result in death of vegetation. Consequently the potential for nitrate leaching is very high in the urine patch.

Most Waikato soils contain allophanic minerals which are known to enhance the nitrification process (Baber, 1978). Apparently, the presence of allophanic minerals is more important for the nitrification process than the actual number of nitrifiers found in the environment (Sarathchandra, 1978). Thus Waikato soils have a greater potential for generating NO_3-N than other New Zealand soils. In the Waikato, moisture deficits are often experienced during summer and autumn. Due to prevailing dry conditions nitrate tends to accumulate in the surface soil during summer. The summer conditions are favourable for high NH_3 volatilisation losses from urine patches which can reduce the accumulation of N in soil. Despite such reduction, due to poor plant N uptake and reduced leaching, the potential for nitrate to accumulate over the summer period is very high. Ground water samples collected in the Hamilton Basin clearly demonstrate that there is high influx of NO_3-N into shallow aquifers during winter (Selvarajah *et al*., 1994). The nitrification rate slows down during winter, but nitrification is not completely inhibited by the cold conditions. The microbial species (nitrifiers) that oxidise NH_4-N to NO_3-N in temperate soils are capable of nitrifying under cold soil conditions. Evidently, NO_3-N production has been observed in moist soils stored under refrigerated conditions (Selvarajah *et al*., 1987).

Steele (1982) estimated that about 110 kg N ha^{-1} y^{-1} is leached in a dairy farm in the Waikato region. Ground water studies performed in the Hamilton Basin by Marshall (1986) from 1981 to 1985 showed average NO_3-N levels of 12.1 g m^{-3} (selected 5-10 m shallow bores) and 5.7 g m^{-3} (> 10 m deep). Selvarajah *et al*. (1994) estimated that approximately 60 kg NO_3-N ha^{-1} y^{-1} could leach annually to shallow ground water in the Hamilton Basin under worst-case conditions (i.e. high soil nitrate accumulation over the summer followed by winter rainfall). A recent field study using ceramic cup samplers at Ruakura showed that the amount of NO_3-N leached could vary depending on the amount of rainfall received (Ledgard *et al*., 1996). The study estimated that 12 kg NO_3-N ha^{-1} leached during a dry year (1993) compared to 74 kg NO^3-N ha^{-1} during a wet year (1994). Considering the high potential for leaching in the Waikato region (high rainfall coupled with relatively high nitrification rates), an estimated 100 kg N ha^{-1} is considered as the maximum annual leaching loss rate for the Waikato region.

Denitrification losses

Denitrification is a process where NO_3-N is reduced to nitrous oxide (N_20) and/or dinitrogen (N_2) under reduced or anaerobic conditions. Generally denitrification losses are greater from urine affected areas of pasture than from urine unaffected areas. This is because in urine patches an initial high availability of organic-C (due to soil pH increase) enhances denitrification of residual NO_3-N, and later, NO_3-N formed from urea-N participates in the denitrification process. Sherlock *et al*. (1992) estimated that under New Zealand conditions N_2O loss from urine-affected pasture was 30 g N_2O-N ha^{-1} d^{-1} and that from urine-unaffected pasture was 10 g N_2O-N ha^{-1} d^{-l}.

It has been general practice to measure the amount of denitrification from the soil surface (topsoil). However, the extent of denitrification below the soil surface and in shallow ground water is several times greater than denitrification from the soil surface. Barkle *et al*. (1993) found that after a regular application of dairy farm effluent on a poorly drained soil (Te Kowhai silt loam) with an established pasture and perched water table, a substantial amount of dairy farm effluent-N applied onto pasture was unaccounted for (354-361 kg N ha^{-1}). These authors suspected that the applied-N unaccounted for must have either been denitrified or stored as organic-N. Their results also showed that there was little or no ammoniacal-N present in the topsoil. These observations suggest that the ammoniacal-N in the dairy farm effluent is nitrified rapidly in the topsoil and either leached and subsequently denitrified or denitrified *insitu*. It is worth noting that the leachate obtained from these soil cores contained a substantial amount of iron.

Under reduced conditions many Waikato soils general considerable amounts of the reduced form of iron (Fe^{2+}) that can also act as an effective electron donor, which is one of the main prerequisites for the denitrification process. It has become a general rule of thumb that ground water aquifers which contain significant levels of iron (>0.2 g m^{-3}) will contain very little or no NO_3-N (Selvarajah *et al*., 1994). Considering the large number of ground water aquifers in the Waikato region with high level iron levels, it is reasonable to assume that subsurface denitrification is one of the most important N removal mechanisms in certain parts of the Waikato region. However, from the point of view of N balancing in a dairy pasture system, the estimation of the subsurface denitrification is less important because N budgeting only requires the extent of leaching loss of NO_3-N from the rhizosphere, not the fate of leached NO_3-N from the soil-plant system. Consequently, it has been decided to use the estimate by Sherlock *et al*. (1992) to determine denitrification losses from grazed pasture.

Assumptions: $N_2O : (N_2 + N_2O) = 0.7$
N_2O loss from urine-affected pasture = 30 g N_2O-N ha^{-1} d^{-1}
N_2O loss from urine-unaffected pasture = 10 g N_2O-N ha^{-1} d^{-1}
Denitrification rate is similar throughout the year from urine-affected and urine-unaffected areas

Calculations: Total ($N_2 + N_2O$) loss from urine-affected pasture = 30 ÷ 0.7 = 42.9 g N ha^{-1} d^{-1}
Total ($N_2 + N_2O$) loss from urine-unaffected pasture = 10 ÷ 0.7 = 14.3 g N ha^{-1} d^{-1}
Total N lost through denitrification = 57.2 g N ha^{-1} d^{-1}
Annual loss through denitrification = 57.2 g N ha^{-1} d^{-1} x 365 d = 20.8 kg N ha^{-1}
= 21 kg N ha^{-1}

Other N pathways

Other N pathways are immobilisation of mineral-N. mineralisation of organic-N, surface runoff of N, and set and dry deposition from the atmosphere. None of these pathways are considered in developing an N budget for dairy pasture system for the following reasons:

(i) *Immobilisation-mineralisation turnover:*

Apart from the mineral-N derived from urine patches, the soil organic-N pool supplies a substantial quantity of mineral-N. Soil organic-N is the largest pool of N in soil (about 6000 kg ha^{-1} in 20 cm topsoil), hence it is difficult to notice any accumulation or depletion occurring in the organic-N pool. The dynamics of organic-N changes in soil are poorly understood. Under New Zealand conditions there may be a depletion of mineralisable organic-N reserves in soil during spring and early summer, however, a clover-based dairy pasture system is capable of replenishing organic-N during cold months of the year (Field and Ball, 1982). Consequently, it is assumed that there is little or no change in net annual N turnover.

(ii) *Surface runoff:*

Surface runoff may occur during high rainfall. The potential for surface runoff is greater for dung-N than for urine-N. Urine-N is likely to leach into the soil profile during high rainfall. Surface runoff of dung-N increases with the increasing slope, rainfall intensity, clay/silt content and soil compaction and decreasing density of vegetative cover. Currently there is no information available on direct measurement of surface runoff of N in a dairy pasture system. Pastoral lands with moderate to steep slopes in the Waikato region are used mainly for sheep and beef farming (eg Waitomo, Coromandel, Taupo and Franklin areas). Most dairy farms are located in the lowlands of the Waikato region and hance the potential for surface runoff loss of N is considered to be minimal in a dairy pasture system in the Waikato region.

(iii) Wet and dry deposition from the atmosphere:

There is insufficient information available on this N pathway in New Zealand. In Europe NH_3 is considered as an atmospheric pollutant (Williams, 1992). It is produced mainly from livestock farms where animals are housed permanently (piggeries) or temporarily (dairy). The main source of NH_3, is from the effluent produced from these farms. In New Zealand the main source of NH_3 is grazed pasture. Although the NH_3 volatilisation loss can be quantified for New Zealand (360000 t y^{-1}) on the assumption of 25% loss from urine-N and 5% loss from dung-N (Hedley *et al*., 1990), it is difficult to quantify the return of this volatilised NH_3 back to pasture. An estimate made on the basis of the above figure indicates that about 14 kg N ha^{-1} should theoretically return on the entire New Zealand land area. This estimate is considered to be conservative, because it does not take into account other sources of NH_3 (e.g. poultry, piggery, meat industry, industrial and municipal areas). Under New Zealand conditions, such predicted values of N deposition do not occur because of prevailing wind directions and turbulence. Total -N measurements for rainwater at Ruakura, Hamilton indicate that the amount of N through wet deposition is very small (2 kg ha^{-1} y^{-1}) (Barkle *et al*., 1993). Thus the input of N to a dairy pasture system through atmospheric deposition is negligible for the Waikato region.

Nitrogen budget

The annual estimate for the consumption of plant-N and the excreta-N flows are illustrated in Figure 1.

Figure 1. Fate of N ingested by cows in a dairy pasture system in the Waikato Region (kg N ha^{-1} y^{-1})

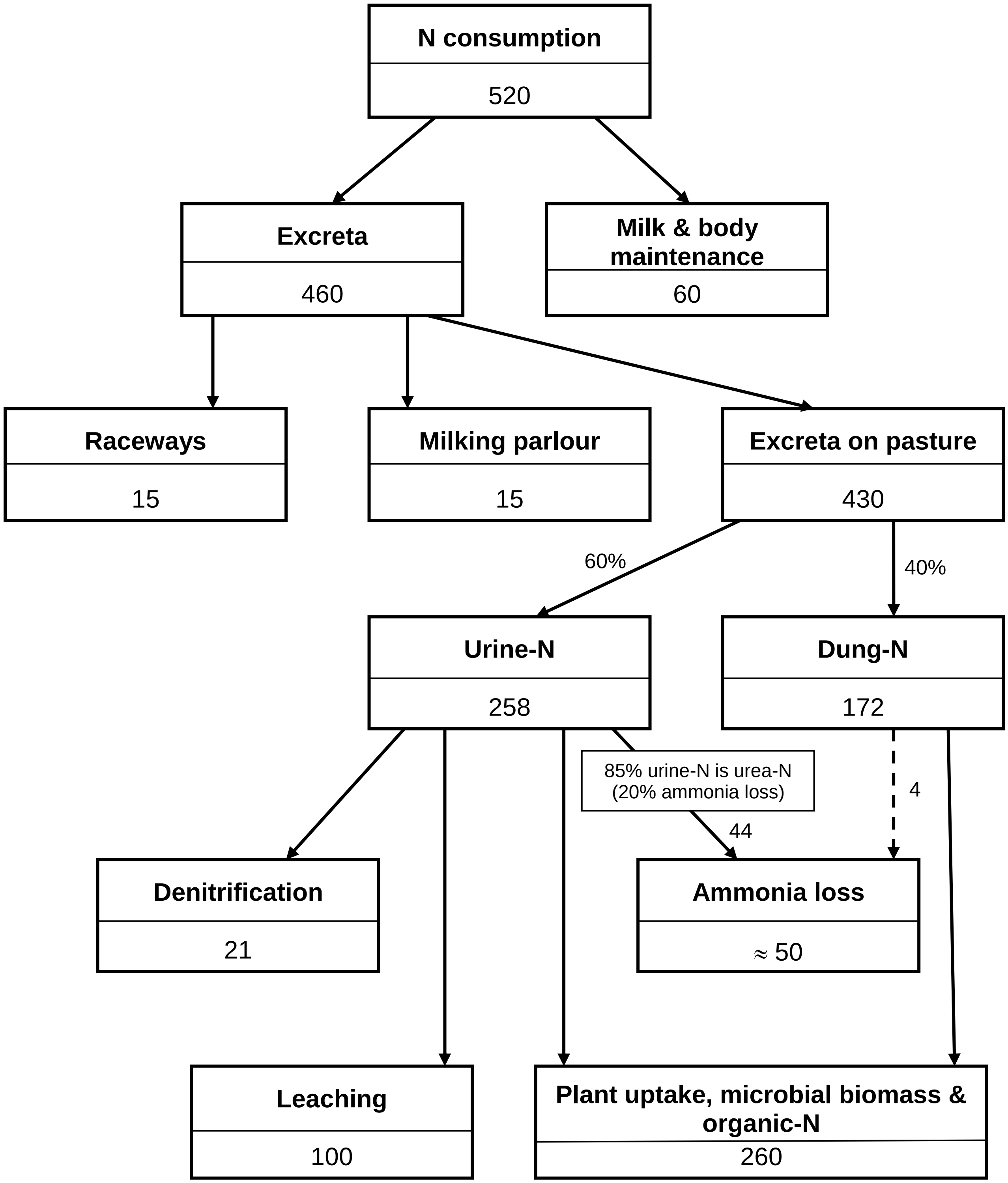

Table 1. N balance for grazed clover-based dairy pasture systems in the Waikato region

N transformation process	Input (kg N ha^{-1} y^{-1})	Output (kg N ha^{-1} y^{-1})
Clover-N fixation	200	-
Products (milk + maintenance)	-	60
Ammonia volatilisation	-	50
Leaching	-	100
Raceways	-	15
Milking parlour	-	15
Denitrification	-	21
TOTAL	200	261

DEFICIT = INPUT-OUTPUT: ≈ -60 kg N ha^{-1}/y^{-1}

The validity of the N balance
It must be emphasised that the N deficit estimated here will be affected significantly by two N transformation processes, i.e. clover-N fixation and leaching losses (than other pathways such as denitrification, NH_3 volatilisation, N transfer to non-productive areas and product removal). Although a denitrification loss of ± 21 kg N ha^{-1} y^{-1} is probable in the Waikato region from well drained soils, the error margin for this estimate is relatively small. The NH_3 volatilisation estimate (i.e. 50 kg N ha^{-1} y^{-1}) is considered to be a maximum estimate for the Waikato region. Thus, generally, the *actual* NH_3 losses sustained in the region are likely to be much less than this estimate. Similar situation applies to leaching losses of N. An average annual loss of 60 kg N ha^{-1} y^{-1} will have more practical implications compared to a maximum estimate of 100 kg N ha^{-1} y^{-1}. In contrast, the clover-N fixation estimate (i.e. 200 kg N ha^{-1} y^{-1}) has smaller error margin and is appropriate throughout the Waikato region under most conditions assuming a clover population of 30% of the pasture. Extreme conditions, for example, severe or prolonged drought conditions could result in poor clover performance hence low N fixation. Such a reduction in N input may not affect the net N balance significantly because similar climatic conditions could also result in a reduction of $N0_3$-N leaching losses. Thus changes in the N input and output could balance each other. In contrast, the drought effects on the magnitude of N removal through milk production or non-productive N transfer are small compared with N leaching. It is estimated that a 25% reduction in milk production due to drought conditions will result in approximately 38 kg N removal as milk protein, which is only 12 kg N reduction in the N output. In short, it could be argued that the N deficit estimated here is considered as the *upper* limit and that often there will be a surplus of mineral-N in a grazed dairy pasture in the Waikato region.

N losses due to effluent irrigation
Effluent can be irrigated from the following sources:

- (i) raw effluent from a sump containing a trip mechanism;
- (ii) raw effluent stored in a temporary storage system for land application; and
- (iii) holding ponds (holding ponds are either specifically designed to hold effluent for a given period or effluent pond systems that have been converted to function as holding systems).

The effluent obtained from all the above sources is referred to as dairy farm effluent and hence the quality of the effluent varies greatly. Dairy farm effluent contains animal excreta, water used for cleaning the floor, milking equipment and milk vats, detergents and soil transferred by animals. Most farms have little or no provision for storm water diversion (collected in the milking area) and this water can runoff with dairy farm effluent.

Detergents are used in the milking parlour mainly to wash milking equipment. The extent and types of detergents used can vary greatly. The detergents used are either acidic or alkaline. Generally acidic detergents are used daily and alkaline detergents are used either on a weekly or monthly basis. The amount of detergent used varies from about 150-300 mL d^{-1}. Depending on the amount and strength of the detergents used, acidic detergents can enhance the preservation of NH3 whilst alkaline conditions can accelerate the NH_3 volatilisation process. Most raw effluent has an elevated pH (8.5-8.9), mainly because of urea-N forming NH_3 (Eq. [1]). Elevated effluent pH is the driving force for the NH_3 volatilisation process or an effluent containing high ammoniacal-N and urea-N, which is the initial stage of the excreta-N loss process. Consequently, any chemical that affects the net pH of effluent can affect the NH3 loss process. A substantial amount of NH_3 can be lost from fresh effluent within two to three days following effluent collection. When the effluent is stored in holding ponds (or anaerobic ponds) for longer periods, N losses can be severe due to ammonification of organic-N. There is insufficient information available on the changes in N content of effluent stored for a given time period.

Effluent containing ammoniacal-N and/or urea can also sustain volatilisation losses when effluent is irrigated onto pasture. Information available in the overseas literature on NH3 loss from surface applied farm effluent is plentiful (Beauchamp *et al.*, 1982; Ryden, 1984; Thompson *et al.*, 1987; Smith and Chambers, 1993). Little or no information is available on the loss pathways of dairy farm effluent N surface applied onto pasture in New Zealand. Ammonia volatilisation from piggery effluent irrigated onto pasture has been estimated as 10% (20 kg N ha^{-1}) of the applied-N (200 kg N ha^{-1}) for a Lismore silt loam under Canterbury conditions (Cameron and Rate, 1992). Piggery effluent contains greater levels of N (0.17% compared with 0.02% in dairy farm effluent) (Vanderholm, 1984) and a very high proportion of ammoniacal-N (about 85% of total-N) (Cameron and Rate, 1992) and therefore has a greater NH_3 volatilisation potential than dairy farm effluent. It has been estimated that trace amounts of NH_3 are lost (5 kg ha^{-1}) from dairy farm effluent (collected from a storage tank) applied onto a pastoral soil (Te Kowhai silt loam) at very high rates (511 kg N ha^{-1}) (Barkle *et al.*, 1993). The potential for NH_3 loss is lower for effluent irrigated from anaerobic dairy ponds due to relatively lower pH. For example Cameron *et al.* (1993) estimated a loss of 3.2 kg NH_3-N ha^{-1} from anaerobic dairy pond effluent irrigated onto pasture under Canterbury conditions. As a conservative estimate 5 kg N ha^{-1} y^{-1} is taken to be NH_3 lost from dairy farm effluent following irrigation onto pasture in the Waikato region.

Due to high dissolved organic-N levels in dairy farm effluent, pasture irrigation of effluent could result in denitrification. Raw dairy farm effluent or effluent from anaerobic dairy pond has no $N0_3$-N and hence denitrification can only occur when effluent is irrigated onto pasture provided NO_3-N is present in soils. Denitrification rates are greater soon after the effluent irrigation and declines rapidly with time. Generally peak losses are sustained within a day of effluent application. Russell *et al.* (1991) showed that the denitrification losses from dairy farm effluent irrigated onto pasture resulted in 1-95 g N ha^{-1} h^{-1} soon after the irrigation and 0.4-1 g N ha^{-1} y^{-1} between irrigation events. Estimated annual denitrification due to effluent irrigation is as follows:

Assumptions: Effluent application frequency = 3 y^{-1}
Peak denitrification period = 1 d irrigation event'
Period of post peak denitrification = 7 d irrigation $event^{-1}$
Denitrification rate is constant
Denitrification rate for peak period = 95 g N ha^{-1} h^{-1}
Denitrification rate for post peak period = 1 g N ha^{-1} h^{-1}

Calculations:
Annual denitrification for peak periods = 3 x 95 g N ha^{-1} h^{-1} x 24 h ÷ 1000 ≈ 7 kg N ha^{-1}
Annual denitrification for post peak periods = 3 x 7 x 1 g N ha^{-1} h^{-1} x 24 h ÷ 1000 ≈ 0.5 kg N ha^{-1}
Total annual denitrification due to effluent irrigation = 7.5 kg N ha^{-1}

Considering the error associated with the above estimate the annual denitrification due to effluent irrigation is rounded to 10 kg N ha^{-1}.

The total-N loss due to irrigation = NH_3 loss-from irrigated effluent + denitrification due to irrigation
= 5 + 10 = 15 kg N ha^{-1} y^{-1}

Annual effluent-N loading for grazed clover-based pasture in the Waikato region

Recommendations for N applications made for *grazed* pasture that are determined solely on the basis of plant uptake of N should be considered with caution. Crop production without the presence of grazing animals (hay or silage making, wheat or barley production) can be managed with significant N inputs - with little or no impact on the environment. In this case, the major N removal pathway is plant uptake of N and the N application can be determined mainly on the basis of crop N removal.

The response of clover has not been quantified for dairy farm effluent applied onto clover-based pasture. However, an indirect assessment of clover N fixation on a Waikato soil (Te Kowhai silt loam) applied with a weekly dose of raw dairy farm effluent (total application of 511 kg ha^{-1}) for 8 months showed that clover can fix about 241 kg N ha^{-1} (Barkle *et al.*, 1993). Consequently, it is assumed that clover-N fixation for grazed pasture irrigated with dairy farm effluent it is comparable with that of pasture without effluent irrigation.

It is not known whether long-term irrigation of dairy farm effluent could alter N dynamics in soil. Unlike industrial land treatment sites dairy farm effluent is irrigated in small quantities with less frequency. Moreover, dairy farm effluent strength is also low compared with other farm and industrial effluents. Thus until further research is performed, it is assumed that the N dynamics for effluent irrigated pasture are comparable with that of grazed pasture with zero effluent irrigation.

Maximum potential annual N deficit in the grazed pasture system before effluent irrigation = 60 kg N ha^{-1}
Maximum potential annual N loss due effluent irrigation = 15 kg N ha^{-1}
Maximum total annual N requirement for effluent irrigated pasture = 75 kg N ha^{-1}

According to the above estimate application of 75 kg N ha^{-1} would help remove the N deficit from the system. As emphasised earlier, this deficit depends heavily on the extent of NO_3-N leaching and NH_3 loss from animal excreta voided onto pasture. Since pastoral soils in the Waikato have a good reserve of organic-N it could be argued that mineral-N could be released from the organic-N pool to reduce or remove the N deficit. However, since agronomical practices avoid risks related to nutrient availability for plants, farmers prefer a nutrient surplus rather than a deficit. To remove the above deficit, at least for the short-term, manure or fertilisers containing readily available N have to be applied. Mineral-N in dairy farm effluent is available as ammoniacal-N. The amount of mineral-N in dairy farm effluent varies depending on the age of the effluent. According to the information collected at Environment Waikato, the amount of mineral-N in raw or treated effluent could vary from 10% to 50% of the total-N (unpublished data). As stated earlier, raw effluent is likely to contain a greater proportion of mineral-N compared with treated effluent. Using the conservative estimate of 50% of total-N as mineral-N, the dairy farm effluent loading requirement to alleviate the N deficit is 150 kg N ha^{-1} y^{-1}. This means that approximately 75 kg N will be available as organic-N.

From an environmental viewpoint, an effluent loading rate of 150 kg N ha^{-1} y^{-1} will have little impact in the short-term. In the long-term, however, more N will accumulate in soil irrigated with effluent and hence the NO_3-N leaching losses could be greater in soils irrigated with effluent. Nevertheless, if the relative environmental risk is considered, the risk of water pollution is small because effluent is treated by soil and the land area used is relatively small. Assuming an effluent loading rate of 150 kg N ha^{-1} y^{-1} and effluent strength of 0.04% N, a dairy a dairy farmer will typically require only 12% of the effective grazed area for irrigation (Environment Waikato, 1994). Despite the anticipated introduction of sophisticated tertiary treatments in the future to treat dairy farm effluent, desired environmental outcomes are easily and effectively achieved with the introduction of land treatment systems. This is because regardless of high quality treatment of dairy farm effluent, discharges into waterways will continue to cause surface water pollution (e.g. increased biochemical oxygen demand, bacterial contamination and nutrient enrichment). Consequently, pasture irrigation of effluent is the best option for treating dairy farm effluent.

A significantly lower loading rate than 150 kg N ha^{-1} y^{-1} could mean larger irrigation area requirements and hence farmers would have to invest more capital in effluent irrigation systems (e g high power pumps, more piping). Moreover, farms in rolling areas will have difficulties finding suitable irrigation areas. Using an effluent loading rate of 150 kg N ha^{-1} y^{-1} is not only considered as environmentally acceptable, but agronomically practicable and sustainable. If environmental risks related to an activity are minimal, the activity can be *permitted* through regional plans by the regional councils. This means that such an activity does not require a resource consent from the local authorities. In contrast, a treated dairy farm effluent discharge into a waterway is considered as a *discretionary* activity by Environment Waikato and hence requires a resource consent including an annual charge. The success of introduction of the dairy farm effluent land treatment systems as a permitted activity in the Waikato region cannot be understated. For example, before the introduction of dairy farm effluent rules in the Waikato region (Environment Waikato, 1994), about 30% of farms irrigated dairy farm effluent whilst the balance used pond or barrier ditch treatment systems to treat the effluent. Currently an estimated 60% of 6000 farms are using dairy farm effluent irrigation systems in the Waikato region.

Conclusions

This paper illustrates that in the Waikato region a well established grazed clover-based pasture system *will not* sustain a chronic mineral-N deficit in the system. This means that dairy farmers in the Waikato region could sustain dairy production without the need for manure or fertiliser-N input provided clover-based pasture is managed effectively.

The information used in this paper to obtain the N balance for a grazed pasture system indicates that more *field* soil-N research is required to fill the existing large information gap on N transformation processes. High quality information will ensure in refining current assessment of components of the N balance.

Considering the worst case of N loss from grazed clover-based pasture systems in the Waikato region low mineral content of dairy farm effluent, and a manageable effluent irrigation area, a recommendation of 150 kg N ha^{-1} annual N loading rate is made for land application of dairy farm effluent.

Since N in the effluent is in both mineral and organic forms (with long-term and short-term mineralisable N) it is an ideal manure for land application. Unlike its mineral counterpart (fertiliser-N), this manure is believed to release N slowly depending on the demand for mineral-N in soil. There is very little information on the effects of dairy farm effluent application on pasture performance and the environment at rates such as 150 kg N ha y . It must be emphasised that only long-term *field* research on land application of dairy farm effluent onto grazed clover-based pasture can reveal the transformation dynamics of soil N in a clover-based pasture system.

Acknowledgements
G R Maggs (Hydrologist, Resource Information Group, Environment Waikato) kindly read the paper and provided helpful comments. Thanks are also due to A Mora (Technical Assistant, Resource Information Group, Environment Waikato) for draughting the original report.

References

Baber, H.L. 1978. A study of some nitrate and phosphate problems in New Zealand agriculture. D. Phil. Thesis, University of Waikato, Hamilton.

Ball, R, Keeney, D.R., Theobald, P.W. and Nes, P. 1979. Nitrogen balance in urine-affected areas of a New Zealand pasture. *Agronomy Journal 71*, 309-314.

Ball, R and Keeney, D.R. 1983. Nitrogen losses from urine-affected areas of a New Zealand pasture under contrasting seasonal conditions. *Proceedings XIV International Grassland Congress*, Lexington, U.S.A. pp 342-344.

Barkle, G., Singleton, P. and Roberts, A. 1993. Progress report on enhancing denitrification of 'land based effluents. **In** "Land Application of Farm Wastes", NZ Land Treatment Collective, Papers for Technical Session No. 9.

Beauchamp, E.G., Kidd, G.E. and Thurtell, G. 1982. Ammonia volatilization from liquid dairy cattle manure in the field. *Canadian Journal of Soil Science* 62, 11-19.

Cameron, K.C. and Rate, A.W. 1992. The fate of nitrogen in pig slurry applied to a stony pasture soil. **In** "The Use of Wastes and Byproducts as Fertilizers and Soil Amendments for Pastures and Crops", (P.E.H. Gregg and L.D. Currie, eds.). Occasional Report No. 6, Fertilizer and Lime Research Centre, Massey University, Palmerston North, pp. 314-326.

Cameron, K.C., Noonan, M.J., Rate, A.W., Condron, L.M, Enright, P.D, Greenwood, P.B. Kerr, L.E., Scott, W.R. and Sherlock, R.R. 1993. The fate of organic wastes injected into and applied to the surface of soil: experimental procedure and initial results. **In** "Land Application of Farm Wastes", NZ Land Treatment Collective, Papers for Technical Session No. 9.

Dairy Statistics. 1995. Livestock Improvement, Private Bag 3016, Hamilton, New Zealand, pp 1-40.

Doak, B.W. 1952. Some chemical changes in the nitrogenous constituents of urine when voided on pasture. *Journal of Agriculture* **42,** 162-171.

Environment Waikato. 1994. Changes to Transitional Regional Plan. 1. Dairy Shed Operative Plan. Environment Waikato Technical Publication Number 1994/5, Environment Waikato, Hamilton, pp 1-31.

Field, T.R.O. and Ball, P. Roger. 1982. Nitrogen balance in an intensively utilised dairy farm system. *Proceedings of the New Zealand Grassland Association* **43,** 64-69.

Goold, G.J. 1980. Rates of dairy shed effluent applied to pastures on clay soils in Northland. *New Zealand Journal of Experimental Agriculture* **8,** 93-99.

Hedley, M.J., Tillman, R.W. and Ball, Roger; P. 1990. Nutrient losses from New Zealand Agriculture. **In** "The Challenges for Fertiliser Research in the 1990's", Proceedings of the New Zealand Fertiliser Manufacturers' Research Association Conference, pp 62-91.

Hoult, E.H. and McGarity, J.W. 1987. The influence of sward mass, defoliation and watering on ammonia volatilization losses from an Italian ryegrass sward topdressed with urea. *Fertilizer Research* 13,199-207.

Jarvis, S.C., Hatch, D.J. and Roberts, D.H. 1989. The effects of grassland management on nitrogen on nitrogen losses from grazed swards through ammonia volatilization; the relationship to excretal N returns from cattle. *Journal of Agricultural Science, Cambridge* **112,** 205-216.

Jarvis, S.C. 1993. Nitrogen cycling and losses from dairy farms. *Soil Use and Management* ***9,*** 99-105.

Ledgard, S.F. 1989. Nitrogen fixation by pasture legumes as influenced by soil or fertilizer nitrogen. **In** "Nitrogen in New Zealand Agriculture and Horticulture" (R.E. White and L.D. Currie eds.), Occasional Report No. 3, Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand, pp 131-138.

Ledgard, S.F., Brier, G.J, Upsdell, M.P. 1990. Effect of clover cultivar on production and nitrogen fixation in clover-ryegrass swards under dairy cow grazing. *New Zealand Journal of Agricultural Research* 33, 243-249.

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

Ledgard, S.F., Penno, J.W., Sprosen, M.S. and Brier, G.J. 1996. Impact of rate of nitrogen fertiliser application on nitrate leaching from grazed dairy pasture. **In** "Recent Developments in Understanding Chemical Movements in Soils: Significance in relation to water quality and efficiency of fertiliser use". Occasional Report No. 9. Fertilizer and Lime Research Centre, Massey University, Palmerston North.

Marshall, T.M. 1986. Groundwater chemistry characteristics of the Mangaonua-Mangaone catchments. Waikato Valley Authority. Technical Report No.86/2. pp 1-23.

Okereke, G.U. and Meints, V.W. 1985. Immediate immobilization of labeled ammonium sulfate and urea nitrogen in soils. *Soil Science* **140,**105-109.

Ryden, J.C. 1984. The flow of nitrogen in grassland. *Proceedings of the Fertiliser Society* **229,** 1-44.

Ryden, J.C., Ball, P.R. and Garwood, E.A. 1984. Nitrate leaching from grassland. *Nature* **311,** 50-53.

Ryden, J.C., Whitehead, D.C., Lockyer, D.R., Thompson, R.B., Skinner, J.H. and Garwood, E.A. 1987. Ammonia emission from grassland and livestock production systems in the UK. *Environmental Pollution* **48,**173-184.

Sarathchandra, S.U. 1978. Nitrification activities of some New Zealand soils and the effect of some clay types on nitrification. *New Zealand Journal of Agricultural Research* **21,** 615-621.

Selvarajah, N., Cameron, K.C. and Swift, R.S. 1987. An evaluation of some laboratory methods of assessing me availability of nitrogen in agricultural soils. *A report to the New Zealand Fertiliser Manufacturer's Research Association Inc., The Ravensdown Fertiliser Co-operative and The Petrochemical Corporation of New Zealand Ltd.,* Department of Soil Science, Lincoln University, Canterbury, New Zealand, pp 1-104.

Selvarajah, N. 1991. Field studies of ammonia volatilisation potentials of unsaturated soils fertilised with urea granules. Ph.D. thesis, Lincoln University, Canterbury, New Zealand, pp 1-182.

Selvarajah, N., Sherlock, R.R. and Cameron, K.C. 1993. Ammonia volatilisation potentials of New Zealand agricultural soils broadcast with granular urea. NZ Land Treatment Collective, Papers for Technical Session No.8.

Selvarajah, N., Maggs, G.R., Crush, J.R. and Ledgard, S.F. 1994. Nitrate in ground water in the Waikato region. **In** "The Efficient Use of Fertilizers in a Changing Environment: Reconciling productivity with sustainability". (Eds. L D Currie and P Loganathan). Occasional Report No. 7. Fertilizer and Lime Research Centre, Massey University, Palmerston North, pp 160-185.

Sherlock, R.R., Muller.C., Russell, J.M. and Haynes, R.J. 1992. Inventory information on nitrous oxide. Report for the Ministry for the Environment, Wellington, pp 1-72.

Sherlock, R.R. and van Der Weerden, T.J. 1992. New methods for measuring ammonia release from urine and ammoniacal wastes applied to pasture. **In** "The Use of Wastes and Byproducts as Fertilizers and Soil Amendments for Pasture and Crops". (Eds. P E

H Gregg and L D Currie). Occasional Report No. 6. Fertilizer and Lime Research Centre, Massey University, Palmerston North, pp 287-296.

Smith, K.A. and Chambers, B.J. 1993. Utilizing the nitrogen content of organic manures on farms - problems and practical solutions. *Soil Use and Management* **9,**105-112.

Steele, K.W. 1982. Nitrogen in grassland soils. **In** Nitrogen Fertilisers in New Zealand Agriculture. Ed. P.B. Lynch. Ray Richards, Auckland, New Zealand, pp 29-44.

Sugimoto, Y. and Ball, P. Roger. 1989. Nitrogen losses from cattle dung. *Proceedings of the XVI International Grassland Congress*, Nice, France.

Thompson, R.B., Ryden, J.C. and Lockyer, D.R. 1987. Fate of nitrogen in cattle slurry following surface application or injection to grassland. *Journal of Soil Science* **38,** 689- 700.

TRC Technical Report, 1990. Review of Monitoring and Inspectorial Procedures for Dairy Shed Oxidation Pond Waste Treatment Systems. Technical Report 90-42. Taranaki Regional Council, Private Bag, Stratford, pp 1-59.

Vanderholm, D.H. 1984. Properties of agricultural wastes. **In** "Agricultural Waste Management", NZAEI Project Report No. 32, 2.1-2.8.

Waghom, G.C. and Barry, T.N. 1987. Pasture as a nutrient source. **In** "Livestock Feeding on Pasture" (A.M. Nicol ed.), pp 21-37. Occasional Publication No. 10, New Zealand Society of Animal Production, Ruakura Agricultural Research Centre, Hamilton, New Zealand.

Wickramasinghe, K.N., Rodgers, G.A. and Jenkinson, D.S. 1985. Transformations of nitrogen fertilizers in soil. *Soil Biology and Biochemistry* 17, 625-630.

Williams, A.J. 1992. Fertilisers and the European Environment - the way ahead. European Fertilizer Manufacturers Association (EFMA), Brussels, pp 1-35.

Whitehead, D.C., Lockyer, D.R. and Raistrick, N. 1989. Volatilization of ammonia from urea applied to soil: Influence of hippuric acid and other constituents of livestock urine. *Soil Biology and Biochemistry* 21, 803-808.