THE EFFECT OF 40 YEARS OF EFFLUENT IRRIGATION ON SOIL AND PASTURE PROPERTIES OF THE LACTOSE NEW ZEALAND LAND TREATMENT SITE

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2000

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A thesis presented in partial fulfilment of the requirements for the degree of Masters of Applied Science, Soil and Earth Sciences Group, Institute of Natural Resources, Massey University

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2000

This thesis is dedicated to the memory of my grandfather,

B.J. Powell.

ABSTRACT

The Lactose New Zealand (LNZ) manufacturing plant, situated in South Taranaki, has an annual average daily effluent output of 1400-1600 m^3, over 11 months of the year. Total effluent loading rates are approximately 5 000 m^3 ha^{-1} yr^{-1}. Effluent composition is extremely variable and characterised by high levels of suspended solids, BOD_5, COD, K, total P and Na and low pH. Land treatment of effluent has been occurring for approximately forty years and currently effluent is irrigated onto three dairy farms in the vicinity of the manufacturing plant.

LNZ has experienced some difficulties in operating and managing both the land treatment system and the dairy farms. Problems have related to the degradation of soil, pasture, surface water and groundwater quality.

This study aims to describe the current status of the land treatment system through characterisation of the soil and pasture resource. Factors examined included soil physical, chemical and biological properties, the quality of pasture and the effect of a grazing event on some soil properties.

The soil type on the three land treatment farms is the Egmont brown loam, characterised by high P fixation and moisture retention, strongly developed soil structure and large microbial communities. Paddocks with varying years of effluent application (0, 9, 14, 30 and 40 years) were used in the characterisation of the soil and pasture resource. A range of soil physical, chemical and biological analyses were undertaken. Chemical soil samples were collected from 5 depths down the soil profile. A grazing trial studied the effect of grazing stock on some soil physical properties shortly after effluent irrigation.

Bulk density values in the topsoil of paddocks irrigated for 40 years are very low (0.52 g cm^{-3}) compared with non-irrigated soils (0.89 g cm^{-3}). Bare patches within paddocks irrigated for 40 years had extremely low bulk density values (0.42 g cm^{-3}). Penetration resistance is significantly lower ($P \leq 0.0001$) on irrigated paddocks compared to non-irrigated paddocks, however aggregate stability levels are similar. Soil water retention and moisture contents of the irrigated soils have increased compared to the non-irrigated soils. Effluent irrigation has had no consistent effect on the infiltration rate of the irrigated paddocks.

The effluent irrigation has resulted in a marked increase in the level of soil chemical fertility. Increases of some nutrient levels have not only occurred in the 0-7.5 cm soil depth, but at lower depths. Total carbon levels have increased slightly in all irrigated study paddocks. The largest increase in total carbon was measured at the 15-30 cm soil depth. This indicates that there was a movement of the soluble and fine particulate carbon added through effluent irrigation down the soil profile. Phosphorus adsorption isotherms demonstrated that effluent irrigation has increased the null point concentration of irrigated soils (30 and 40 years) from 1-2 $\mu g\ ml^{-1}$ to approximately 100 $\mu g\ ml^{-1}$, indicating phosphate desorption is likely to occur at shallow depths. Soil solution extract and suction cup solution analysed for P showed that up to an Olsen P level of 130 $\mu g\ cm^{-3}$ of soil, P was retained strongly in the soil and there were negligible concentrations of P in the soil solution. Above this, P concentrations in soil solution were significantly higher enhancing the potential for leaching losses. However, below 30 cm depth, effluent irrigation has had little effect on soil solution P, indicating these soils have the ability to adsorb additional phosphate at depth. Exchangeable sodium and potassium levels have increased significantly ($P \leq 0.05$) on irrigated soils compared with non-irrigated soils at 0-7.5 cm depth. This may lead to soil physical deterioration.

Effluent irrigation has had little influence on earthworm populations and soil respiration rates.

The extensive use of all three farms for effluent irrigation has impacted on shallow groundwater to varying degrees. Interpretation of Taranaki Regional Council data indicated that sodium, nitrate, conductivity and occasionally filtered COD levels in the groundwater of most of the impact bores had increased. However, the groundwater is not used for a potable supply and since the introduction of new management philosophies in 1995, groundwater nitrate levels have decreased.

Irrigated paddocks have similar values for most pasture quality parameters analysed. There is considerable difference between the dietary cation-anion difference (DCAD) of the irrigated and non-irrigated paddocks. Animal diets with high DCAD values tend to increase the incidence of milk fever mainly due to Ca deficiency. This suggests that there is a need to supplement the Ca levels in the forage. Nutrient status of the pasture on the irrigation farms is in the normal to high range recommended by AgResearch for adequate pasture nutrition.

The grazing trial showed grazing of pasture shortly after effluent irrigation had little effect on soil physical properties, however soil moisture levels were low during the trial.

The study has identified some trends that can be used to estimate the rate of change in soil properties due to the addition of effluent under the present conditions. There is the risk of a further decrease in soil bulk density, combined with increased soil moisture content, resulting in treading damage by grazing animals. The soil system still has a very large capacity to fix phosphate. However, of some concern is the likelihood of surface runoff containing high concentrations of P as a result of high levels of available P in the top 0-7.5 cm. The current policy of planting riparian strips should reduce this environmental threat.

The study suggests that forty years of effluent irrigation has had a considerable effect on soil, pasture and groundwater and surface water quality at the LNZ site. The system will require careful management to ensure the sustainable land treatment of LNZ effluent. Alternative management options are discussed.

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CHAPTER 1

Introduction

1.1 Introduction to the company

Lactose New Zealand (LNZ) was established in 1913 as the 'New Zealand Sugar of Milk and Casein Company' Ltd., based at Edendale in Southland. Although the company expanded to a newly built site at Kapuni, Taranaki in 1946, (Figure 1.1) the original site is still occupied by a lactose plant, operated by the New Zealand Dairy Co-operative Dairy Company Ltd.

Ownership of the company was based in the United Kingdom (with the Unigate Company) until 1983, when it was sold to the New Zealand Dairy Board. From its modest beginnings, production has risen to over 37 000 tonnes of product per year. Pharmaceutical and food grades of lactose are now produced by the company and are used in a wide range of products including: infant formula, dairy products, coffee whiteners, artificial sweeteners and confectionary.

The company is currently undertaking a major expansion programme of buildings and facilities to enable LNZ to increase production, make new product grades and further improve the quality of its Wyndale brand lactose (http://www.lactose.co.nz).

The LNZ processing plant is situated at Kapuni, approximately 15 km north of Hawera. The site is located in the heart of Taranaki's intensive dairy land. The Company exports to over 40 countries and employs 109 staff (Pers. comm., Warren Climo 2000). Annual turnover is approximately $55-65 million (Pers. comm., Warren Climo 2000), making LNZ the world's largest 'stand-alone' and third largest total producer of lactose in the world.

Figure 1.1 *The Lactose New Zealand factory set in the fertile countryside of South Taranaki with Mount Taranaki/Egmont in the background.*

LNZ collects whey permeate from Kiwi Dairies (situated in South Taranaki) and Anchor Products' Hautapu (situated in Waikato) dairy factories in the North Island. Whey permeate is a by-product of whey protein concentrate manufacture which uses whole whey derived from casein and cheese manufacture. The whey permeate contains lactose, minerals, residual organic compounds and fragments of protein.

The LNZ manufacturing plant generates up to 2500 m^3 of effluent a day, over 11 months of the year. The annual average daily output is approximately 1400-1600 m^3. The effluent consists of evaporator condensate (water evaporated from the raw materials), plant washings, processing wastes (e.g. spills within the factory), and stormwater from areas that have a potential to be contaminated by product (LNZ, 1998). The effluent is collected in two main sumps and pumped into a 680 m^3 balance tank, prior to pumping to the land treatment system.

1.2 History of effluent treatment

In the beginning, the effluent from the LNZ plant was discharged into the Kaupokonui River, as was common practice for dairy companies at that time. Due to the deterioration of water quality in the river, the company decided to irrigate the effluent onto farmland. Land treatment commenced in 1957, with the irrigation of effluent onto 45 ha of freehold (the farm that is now called the No. 2 farm) and 10 ha of leasehold land (Dryden, 1993), situated 1 km from the factory. Ten years later, the company bought a 79 ha farm (the current No. 1 farm) adjacent to the factory. Approximately half the area of this farm was used for effluent irrigation. Effluent was sprayed on both properties using large sprinklers mounted on trolleys. Two sprinklers were used to irrigate effluent at any one time. Irrigation on any one area was for 4 hours in wet and 10 hours in dry conditions (irrigation occurred for 16-17 hours per day) (NZDRI 1993).

Prior to 1975, the No. 2 farm was used as a 'run-off' for the No.1 farm. In the 1974-75 season, the properties grazed 300 cows and produced 45,300 kgs of milkfat (NZDRI, 1993). In the 1975-1976 season, the two properties were farmed individually, each with their own herd and sharemilker. The irrigated effluent volume was 1100–1400 $m^3\ d^{-1}$ (NZDRI, 1993), equivalent to 650-850 mm yr^{-1}. In September 1984, based on advice from the New Zealand Dairy Research Institute (NZDRI), the sprinkler system was upgraded to a travelling irrigator system and reticulation on the No. 1 farm extended. By now, the effluent volume for irrigation was approximately 2000 $m^3\ d^{-1}$ (NZDRI, 1993), equivalent to 900-1000 mm yr^{-1}.

In 1988, the land treatment system was extended to include the current No. 3 farm. At that stage, the No.3 farm was privately owned and irrigation was undertaken on a contractual arrangement. Hydraulic loadings to the No. 3 farm were lower than farms No. 1 and No. 2 (Pers. comm., Warren Climo 2000) but it was not possible to quantify the volumes applied to the No.3 farm (NZDRI, 1993). The No.3 farm was acquired by LNZ in June 1997 and effluent hydraulic loadings and farm management were made similar to those used on farms No. 1 and No. 2.

1.3 Current land treatment system

The current land treatment system irrigates effluent onto three dairy farms in the vicinity of the manufacturing plant. Figure 1.2 shows the location of all the farms in relation to the factory. All farms are situated on Egmont orthic allophanic soils. Figure 1.3 is a digital photo of a soil profile taken from paddock 24 (non-irrigated) on farm No. 3. The Egmont soils are classic allophanic soils, with high allophane content (10-15%), high porosity (70%), friable consistence and high plant-available moisture retention (approximately 80 mm of readily available water and 200 mm of total available water in the first 1 m of the soil profile) (Molloy, 1993).

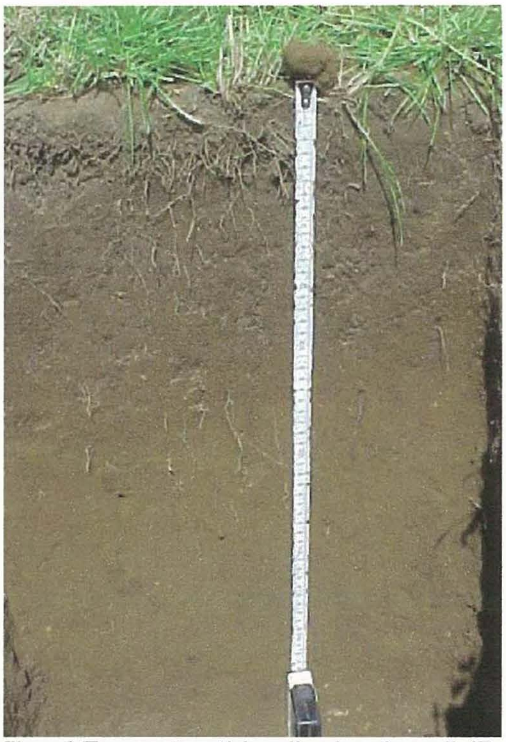


Figure 1.3 *Soil profile of Egmont orthic allophanic soil (Farm No.3, paddock 24)*

The No.1 farm is located immediately to the north of the factory. It is 79 ha in area, with approximately 230 milking cows. The No.2 farm is located on Skeet Road, approximately 1 km, south east of the factory. It has an area of 45 ha and approximately 145 milking cows. The No.3 farm consists of 60 ha, carrying approximately 210 milking cows and is located on the Manaia Road, approximately 2 km south of the factory. The stocking rate of the farms is 2.9 cows ha^{-1} (No.1 farm), 3.2 cows ha^{-1} (No.2 farm) and 3.5 cows ha^{-1} (No.3 farm). The south Taranaki average is 2.9 cows ha^{-1} (LIC, 1999). However, the stocking rate in the Kapuni area is likely to be higher than this.

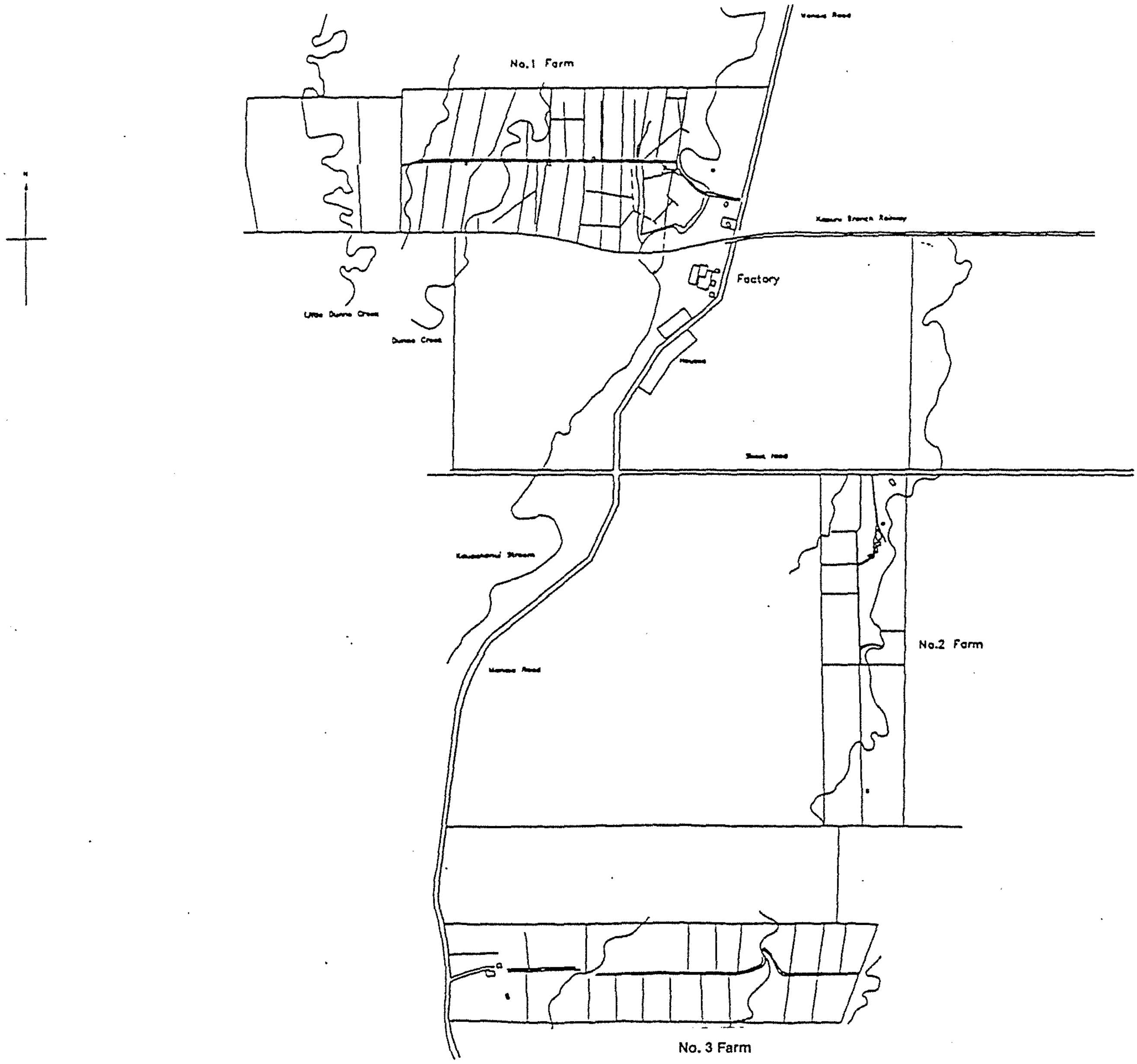


Figure 1.2 *Site diagram for Lactose New Zealand showing the location of the processing plant and dairy farms*

The land treatment system includes, both 'big gun' travelling irrigators (seven Southern Cross TCD2000 irrigators with an average effluent application rate of 14mm hr^{-1}), applying effluent to 95 ha, and 'in-ground sprinklers' (on farms No.1 and No. 2), irrigating 25 ha (Pers. comm., Warren Climo 2000). Figure 1.4 shows both the travelling and in-ground irrigators applying effluent to paddocks. A total of approximately 600 000 m^3 of effluent is applied annually (Pers. comm., Warren Climo 2000), which equates to approximately 500 mm of irrigation over the 120 ha of land currently used for land treatment. Individual applications range from 200-450 m^3 ha^{-1}, which equates to 20-45 mm per irrigation event. The large range in application rates is a reflection of the fluctuation in effluent production throughout the year.

Figure 1.4 *In-ground (top) and travelling irrigators (bottom) in operation*

Under normal conditions, two travelling irrigators are used at any one time on each farm. A travelling irrigator takes approximately six hours to complete a run the length of a paddock. A return cycle of not less than 16 days is used. Each farm receives effluent once every three days but may also receive some effluent in between, when effluent outputs are high. Travelling irrigators are shifted by the sharemilkers who also determine the paddocks to be irrigated. Two employees from LNZ are responsible for the operation and maintenance of the inground sprinklers.

1.4 Historical problems associated with the land treatment system

LNZ has experienced some difficulties in operating and managing both the land treatment system and the dairy farms. Problems relate to the degradation of soil, pasture, surface water and groundwater quality. Large application rates and hydraulic loads increase the likelihood of ponding and surface runoff to waterways. With daily irrigation necessary, at some stages during the year there is the risk that saturated paddocks will be irrigated with effluent. The probability of extended periods of anaerobic conditions under saturated paddocks is also increased, along with the susceptibility of nutrients to leaching. Stock grazing on saturated soil may cause pugging and deterioration of soil structure, leading to soil compaction and decreased infiltration rates.

A 1984 New Zealand Dairy Research Institute report stated that the problems experienced by LNZ could be summarised as (NZDRI, 1984):

- Excessive application rates – reported to be up to 60 mm hr^{-1}.
- Excessive application volumes – reported to be up to 90mm per irrigation event.
- Insufficient land area for irrigation.

The Taranaki Regional Council (TRC) has prepared annual reports on the operations of LNZ for at least 13 years. Included in these reports are observations of the condition of the land treatment system. TRC has stated that "historically the company does not have a very good record relating to its spray irrigation. A number of unauthorised discharges resulting from irrigator breakage or malfunctions have occurred" (TRC, 1994; TRC, 1998). The TRC reports that ponded areas of irrigated wastes have been observed on farms No. 1 and No. 2 during wet periods (TRC, 1995; TRC, 1996). The Council has observed runoff on farm No.1 (TRC, 1988; TRC, 1994; TRC, 1995). Incidents of spray drift into the Motumate Stream have also been reported (TRC, 1995; TRC, 1997). The latest Regional Council report indicates less frequent observations of unauthorised discharges, ponding and runoff of effluent and spray drift into streams (TRC, 1999). However, incidents such as these are sometimes unavoidable and will occur from time to time.

The 1993 NZDRI report noted that some of the problems that have occurred with the land treatment system in recent years relate to changes in staff on the farms and the lack of coordination between LNZ's manufacturing arm and the effluent treatment operation (NZDRI, 1993). LNZ have made a significant effort to co-ordinate all sections of their operation. Current farm staff have been employed for several years and are very experienced in operating the irrigators.

1.5 Current management goals of the land treatment system

In the last five years, LNZ has endeavoured to improve the operation and management of its land treatment system. The management philosophy on the dairy farms was changed in 1995 to give the greatest priority to activities that maintain and improve the farms for sustainable effluent treatment. The financial goals associated with milk production have become a secondary focus.

The following practices were adopted in 1995:

- Removal of all stock from the farms for the winter period (i.e. from the end of the milking season to 2 weeks prior to calving) in order to reduce grazing pressure on soils and pastures over the wetter winter months.

- A pasture renewal programme (via a crop of turnips) to improve the quality and quantity of the ground cover, increase the soil-plant system's ability to withstand grazing and reduce the potential for surface runoff. Generally, at least one paddock per farm is regrassed annually.
- Regular subsoiling of paddocks to increase infiltration rates, aeration, and the ability of the soils to process effluent.
- Liming of paddocks with 1 MT ha^{-1} $year^{-1}$ (first undertaken in 1996) to maintain sustainable mineral balances and reduce the impacts of sodium and potassium additions from the effluent on soil physical and chemical properties.
- Annual soil sampling by the NZDRI to monitor nutrient levels in the soil.

Under the revised management plan there have been annual increases in production on farms No. 1 and No. 2. Table 1.1 shows these increases in comparison with the South Taranaki average.

Table 1.1 Changes in farm production under revised management plan

Season	Farm No. 1		Farm No. 2		SouthTaranaki
	(total ms)	(kg ms ha ha^{-1})	(total ms)	(kg ms ha^{-1})	Average (kg ms ha^{-1})
1994/95	54 679	781	45 893	1 147	755
1995/96	68 337	976	46 956	1 174	744
1996/97	80 816	1 155	53 055	1 326	816
1997/98	87 083	1 244	55 898	1 397	875
1998/99	76 131	1 088	50 254	1 256	786

ms – milksolids

The data indicate that given the difficulties associated with operating a large scale land treatment system within a dairy farm, the farms No. 1 and No. 2 achieve good production. Farms in the Kapuni area are likely to have greater production than the South Taranaki average.

1.6 Climate and environmental setting

The climate of South Taranaki has a major influence on the management of the land treatment system and the dairy farms. The climate is influenced by Mount Egmont/Taranaki, which is located approximately 20 kilometres to the north of the site. Rainfall is moderate to high (mean annual rainfall of 1200 mm) with frequent heavy winter rainfall events. Climate is discussed further in section 2.3.

The LNZ land treatment system is located in an environmentally sensitive area. The Egmont orthic allophanic soil is moderately permeable and therefore leaching of applied nutrients, particularly nitrogen, sodium and potassium to groundwater is a concern. Although groundwater is currently not widely used for drinking and stock water, it is a water resource that could conceivably be used in the foreseeable future.

All three farms have at least one stream dissecting the farm with farm No.1 having 4 streams running through the property. All waterways are freshwater streams originating from Mount Egmont/Taranaki or the surrounding ring plain. Some of the streams are used for gathering food by the local Maori (TRC, 1993) and are also tributaries of the Tasman Sea (Kaupokonui Beach), which is used for food gathering and recreational use. While the small unnamed tributary of the Kaupokonui River, that runs past the oxidation ponds on farm No. 1, has a constructed wetland (shown in Figure 1.5) on its banks, all other streams have had very little riparian vegetation to prevent runoff of effluent to surface water. Some streams are not fenced off, allowing stock to graze stream banks, drink and walk through the water. The company has a riparian planting regime, which has been progressing for the last three years, which is addressing this situation. The gully areas in the vicinity of the farm No.1 (Figure 1.5) cowshed to the lower farm boundary and alongside the Waiokura Stream on farm No.2, from the central race to the lower farm boundary were planted in previous winters and continue to be planted and maintained.

Figure 1.5 *Constructed wetland on unnamed tributary of Kaupokonui River, Farm No. 1*

1.7 Objectives of the investigation

Lactose New Zealand would like to maintain and improve the farms' long-term ability to treat effluent from the factory. To achieve this, LNZ require a description of the current status of the land treatment system. The description includes the characterisation of the soil and pasture resource, which forms the overall objectives of the present investigation. Soil physical, chemical and biological sampling will yield information on soil nutrient levels, the condition of soil structure and strength, and micro-organism activity in the soil.

The specific objectives of the study include:

1. To examine the physical characteristics of the soil (soil strength, bulk density, aggregate stability, soil infiltration rates, soil moisture contents, moisture retentivity).
2. To examine the chemical characteristics of the soil (soil pH, carbon, nitrogen, phosphorus, phosphate retention, cation exchange capacity).
3. To determine soil solution phosphorus concentrations and to identify whether P leaching is likely to be occurring.

4. To examine the biological characteristics of the soil (earthworm populations, soil respiration rates).
5. To assess the effect of grazing animals on soils shortly after effluent irrigation.
6. To characterise some chemical and physical soil characteristics of the bare patches in paddocks on farm No. 2.
7. To examine the quality of pasture.
8. To review groundwater and surface water quality data collected by the Taranaki Regional Council.

This will give an indication of how effluent application has impacted on soil and pasture quality, at different time intervals over a 40 year period. In addition to this, LNZ would like to gauge the sustainability of their land treatment system. This should indicate likely scenarios for pasture, soil, and water quality under present conditions and practices as well as alternatives to the current practices. An overview of the structure of the thesis is presented in Figure 1.6

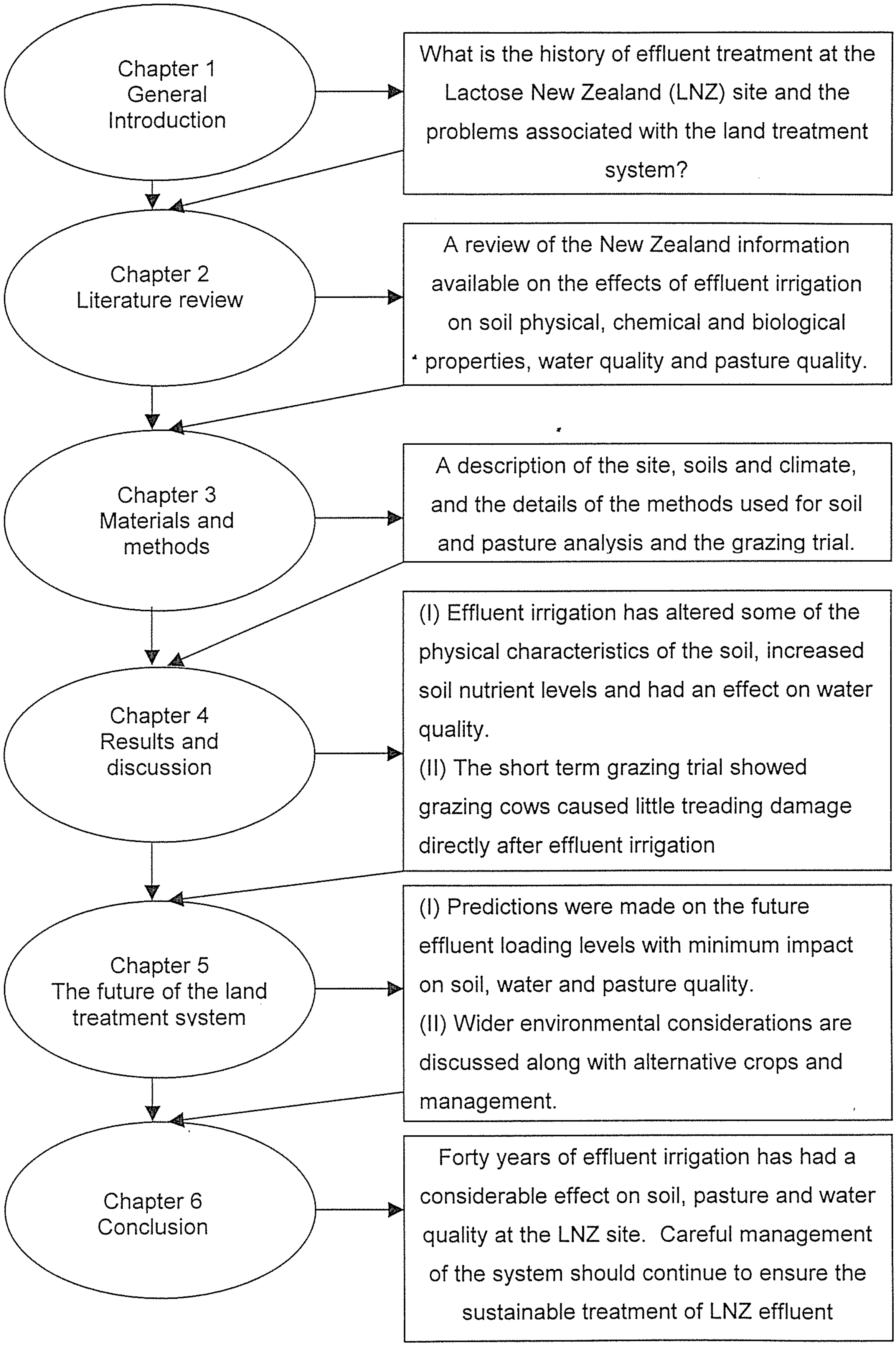


Figure 1.6 *An overview of the structure of the thesis*

CHAPTER 2

Literature Review

2.1 Introduction

This chapter reviews the information available on the effects of effluent irrigation on soil physical, chemical and biological properties. New Zealand literature is predominantly cited in this chapter along with examples from other countries that are appropriate. This was done because New Zealand's unique combinations of soils, effluent, climate and land use patterns, make most overseas situations difficult to make comparisons with. There is a broad overview of a large range of soil properties rather than a detailed critique of more specific areas. This is in accordance with the research undertaken at the Lactose New Zealand irrigation farms.

In New Zealand and many other countries, land application of agricultural, industrial and municipal effluent is being favoured over disposal in waterways (Cameron *et al.*, 1997). Few studies have examined the effects of long-term effluent irrigation on soil quality (Degens *et al.*, 2000).

2.2 Land treatment of manufacturing site effluent in New Zealand

Land application of wastes is becoming more widespread as regulatory authorities move to protect water quality by restricting waste disposal into surface water that includes rivers, lakes and the marine environment (Cameron *et al.*, 1997; Bond, 1998). Spray irrigation of effluent is an important method of effluent treatment in New Zealand, with half the manufacturing sites using this means of treatment.

The irrigation of effluent to farmland has been carried out by the dairy industry for over fifty years with approximately 2 500 hectares of land currently being spray irrigated with effluent (NZDRI, 1993). Low strength organic effluent is sprayed onto farms, which are generally operated as dairy or dry stock units. The majority are dairy farms and are often the highest producing farms in their area. These farms benefit from the nutrients and water applied through the effluent. Careful management of these irrigation farms is important, particularly during spring when the rainfall is high (NZDRI, 1993).

2.3 Effluent characteristics

Some examples of the composition of commonly irrigated wastes in New Zealand are given in Table 2.1.

Table 2.1 Comparison of LNZ effluent with sludges/effluents from a few selected waste sources in New Zealand (Hart and Speir, 1992; Carnus, 1994)

Units are in gm^{-3} except for pulp and paper sludges ($mg\ kg^{-1}$). Dashes, not determined

Parameter	Milk powder/ butter factory waste	Meat processing secondary effluent	Tannery secondary effluent	Pulp and paper secondary sludges	Dairy shed effluent	Piggery effluent	Domestic effluent	Lactose New Zealand effluent
SS	-	20-100	120	-	-	-		360-580
BOD_5	1500	20-100	30	-	-	-		7200-19100
COD	-	80-400	410	-	-	-		12100-28700
pH	10-12	-	7.6	-	-	-		4.4
Total N	70	40-200	130	32000	190	1300	20	60-125
Total P	35	5-30	1.6	8075	30	600	7	70-100
Fat	400	0-30	-	-	-	-		-
Na	560	50-250	2700	4586	50	-	40	80-180
K	13	20-150	-	2905	220	500	10	220-370
Ca	8	3-250	340	17000	110	-	12	-
Mg	1	3-10	36	2000	30	-	3	-

The wide range of chemical, physical, and biological characteristics of these wastes makes it difficult to develop guidelines for their use. The chemical composition not only varies between the waste streams but individual waste streams also vary over time (Cameron *et al.*, 1997).

While the nutrients contained in these wastes make them attractive as fertilisers, their application on land may be constrained by the presence of toxic metals, toxic organics, excessive concentrations of salt, or extreme pH. For example, effluent from dairy, tannery and pulp and paper factories often contains high concentrations of sodium ions that can cause deflocculation of clay particles and breakdown of soil structure (Lieffering and McLay, 1995; Cameron *et al.*, 1997).

The composition of LNZ effluent has changed over time as improved technology has enabled more lactose to be extracted from the whey, washing chemicals to be altered (reducing the effluent pH from alkaline to acid), and carbon to be introduced to the filtering process.

The volume of LNZ effluent varies throughout the year depending on the raw material supply. Little or no effluent is produced in the winter months of June and July. There is extreme variability over short time intervals for all commonly measured effluent properties (NZDRI, 1993).

The key components of the lactose effluent are low pH and high suspended solids, phosphorus, BOD_5 and COD (Table 2.1). The characteristics of LNZ effluent are somewhat different to any other effluent irrigated in New Zealand. Therefore, there is no experience in irrigating effluent with these properties and the long-term effect of irrigating LNZ effluent or other effluents, for that matter, on soil properties is largely unknown.

2.4 Effect of effluent on soil physical properties

Compared with the number of reports on the effects on soil chemistry, there have been few reports on the effects of land application of wastes on soil physical conditions. Land application of wastes can have a variety of beneficial or detrimental effects on soil physical conditions, depending on the characteristics of the waste and the soil (Cameron *et al.*, 1997), the land use and the climate.

2.4.1 Soil moisture content

Soil water holding capacity can be increased by the input of organic matter in the applied waste and by the improved soil physical conditions (Mbagwu and Piccolo, 1990).

Sewage sludge, compost, meat processing plant effluent, and manure additions to soil have all been reported to increase soil water retention (Gupta *et al.*, 1977; Kladivko and Nelson 1979; Khaleel *et al.*, 1981; Ross *et al.*, 1982). This was attributed to increased aggregation and therefore increased total pore space (Khaleel *et al.*, 1981). Degens *et al.* (2000) noted at all depths sampled, soil moisture content was greater under pasture irrigated with a dairy factory effluent for 22 years than non-irrigated pasture.

The application of pig slurry, sewage sludge and cattle slurry is reported to have increased the amount of water retained at a matric potential of –30 kPa (Mbagwu and Piccolo, 1990). The application of poultry manure at a rate of 110 t ha^{-1} for 5 years increased water retention from 32 to 42% at a matric potential of –10 kPa (Weil and Kroontje, 1979).

Pagliai *et al.* (1985) and Pagliai and Antisari (1993) reported an increase in soil porosity following the application of pig slurry and that most of the increase was due to the development of soil macropores (>50 um). The increase in soil macropores is particularly important because of their dominant effect on root growth, water infiltration and aeration.

2.4.2 Soil bulk density

Additions of organic wastes to soil generally cause an increase in soil organic matter and therefore a decrease in soil bulk density. Khaleel *et al.* (1981) derived a linear regression equation (r^2=0.69) between increases in soil organic carbon and reported decreases in soil bulk density. They attributed the decrease in bulk density to the dilution effect resulting from the mixing of the added organic matter. An increase in aggregation and therefore an increase in porosity would also be expected to have contributed to the decrease in soil bulk density.

Obi and Ebo (1995) reported a reduction in bulk density from 1.54 to 1.40 g cm^{-3} and an associated increase in total porosity following the land application of chicken manure at 10 t ha^{-1}. A reduction in bulk density from 1.1 to 0.8 g cm^{-3} was also reported after 5 years of application of poultry manure at 110 t ha^{-1} (Weil and Kroontje, 1979). In most cases where decreases in bulk density have been noticed, large quantities of solid waste have been added.

In New Zealand, McAuliffe (1978) found that bulk density decreased at some whey treatment sites and not others. Marris (1992) found no difference in soil bulk density for two soils to which whey had been applied at fertiliser rates, in the Waikato region.

Decreased bulk density may have a number of site management implications. Water storage capacity in the surface layer would be higher, lessening the likelihood of surface runoff. Soil infiltration rates may increase due to the increase in soil porosity (Hillel, 1980; Pagliai *et al.*, 1985; Pagliai and Antisari, 1993). A low bulk density indicates a lower bearing strength, hence pugging damage during wet periods may become more pronounced (McAuliffe, 1978).

2.4.3 Soil infiltration rates

The rate at which water can enter the soil surface is called the 'soil infiltration rate'. A soil's infiltration rate is an important parameter for determining effluent application rates, to ensure all irrigated effluent infiltrates into the soil. Maintaining soil infiltration rates is essential for the continued viability of an effluent land treatment system. The infiltration rate is often influenced by physical, chemical and biological factors. Some of the physical factors include soil texture, soil structure, bulk density, aggregation, organic matter content, and the types of clay present.

Addition of organic matter to soils is generally expected to increase soil permeability, as it contributes to increased porosity and structural stability (Khaleel *et al.*, 1981; Reed and Crites, 1984). However, land application of wastes is reported to have a variable effect on the infiltration rate and hydraulic conductivity of soils (Cameron *et al.*, 1997).

Increases in soil permeability have been reported following additions of solid wastes (Khaleel *et al.*, 1981), and for effluent additions (Mathan, 1994). Obi and Ebo (1995) reported a 4-fold increase in hydraulic conductivity from 34.8 to 187 mm h^{-1} following the addition of poultry manure. Similarly, Magesan *et al.* (1996) reported that the application of secondary-treated sewage effluent increased macroporosity of a sandy loam from 11 to 19% and the saturated hydraulic conductivity from 39 to 57 mm h^{-1}.

Reductions in infiltration rate have also been reported following the application of some wastes (Lance *et al.*, 1980; Wilcock, 1984; Clanton and Slack, 1987, Cook *et al.*, 1994). For example, decreases in soil infiltration rates were reported with the addition of sewage (Reneau *et al.*, 1989) and fellmongery effluent (Balks and McLay, 1996). McAuliffe (1984) found that soil permeability was reduced by 95% within 2 days of applying wool scour effluent. Reductions in soil permeability have been reported at dairy factory effluent sites in New Zealand (Phillips, 1971; Parkin and Marshall, 1976; McAuliffe, 1978) and in the United States (USEPA, 1971)

In laboratory studies using undisturbed soil cores, McAuliffe *et al.* (1982) found that a single application of simulated whey effluent resulted in approximately a 50% decrease in soil permeability within two days. Repetitive doses of effluent led to a decrease in soil permeability of over 99% in some cores. The decline was suggested to be due to a combination of both physical and biological blockage processes. Cores, which received a single application of simulated whey effluent, followed by water every second day, exhibited virtually, complete recovery within two weeks. Some cores receiving repetitive applications of simulated effluent showed no signs of recovery within 28 days of final application. Soil permeability increases in the soil cores appeared to be the result of the removal of pore blockage, creation of new macropores and enhanced earthworm activity (McAuliffe, 1978).

The main cause of decreasing permeability is the blockage of soil pores. Factors that result in pore blockage may be classified as physical, biological and chemical (Berend, 1967; Rice, 1974). As effluent moves through soil pores, suspended solids are removed and can at times clog soil pores, thereby causing a decrease in the infiltration rate (De Vries, 1972; Bouwer and Chaney, 1974; Rice, 1974; USEPA, 1976; Loehr *et al.*, 1979; Reed and Crites, 1984; Vandevivere and Baveye, 1992). Balks (1995) reported the primary cause of declining infiltration rates, identified at meat processing land treatment sites, was suspended solids in the effluent. Generally, in all but coarse sands and gravel, the suspended solids are filtered out at, or very close to the soil surface, where they tend to form a thin layer of crust with low permeability that may occur as an immediate result of effluent addition (Balks, 1995). Vinten *et al.* (1983) observed that in a silt loam soil, the coarse solids from sewage effluent were filtered out in the top 10 mm of soil, while below this level finer solids were deposited by adsorption. The factors influencing physical soil pore blockage include, effluent suspended solids content, the particle size distribution of the solids in the effluent, volume of effluent applied and the pore size distribution of the soil. Development of a surface layer of organic material does not always cause a decline in infiltration rates (Vanderholm and Beer, 1970).

Biological clogging occurs when bacterial growth or its by-products decrease pore diameter. A change in the pore geometry of a soil would have a definite effect on the hydraulic conductivity (Frankenberger *et al.,* 1979). It usually occurs at the soil surface, although it can be at any depth (Thomas *et al.*, 1966; Bouma *et al.*, 1972; Bouwer and Chaney, 1974; Rice, 1974; Spyridakis and Welch, 1976; Kristiansen, 1981) and is a function of the waste organic material added and the soil pore space (Loehr *et al.*, 1979). Biological activity may also lead to pore blockage as a result of production of gases, or due to the microbial destruction of materials that bind soil aggregates, therefore leading to disaggregation (Miyazaki, 1993). The factors that affect the development of this form of blockage include, effluent properties (such as nitrogen, carbohydrates, C:N ratio, phosphorus, other nutrient content, pH, and volume added), temperature, soil moisture content, time between irrigations, and the microbial species inherent in the soil or added in the effluent (Balks, 1995)

Chemical phenomena that affect soil permeability involve changes in the swelling properties of soils, and the dispersion of colloidal particles, brought about by the addition of ions, particularly sodium and other monovalent cations (Vandevivere and Baveye, 1992). Solubilisation of organic matter resulting from the application of alkaline effluent can also lead to a decrease in infiltration rates (Lieffering and McLay, 1995). Factors affecting dispersion are, the soil clay content and mineralogy, the sodium adsorption ratio of the applied effluent, the volume and frequency of effluent applications and the volume and frequency of rainfall that will leach excess salts from the soil. Deflocculation of clay minerals leads to a decrease in soil permeability, poor soil aeration and difficulty in seedling emergence (USEPA, 1981; Reed and Crites, 1984).

A reduction in infiltration rate can cause ponding of effluent to occur and this can increase the odour emitted from some effluent treatment areas. There is also an increased risk of surface runoff occurring, which may cause contamination of nearby waterways. If these problems become serious, they may threaten the viability of the effluent treatment operation (Cameron *et al.*, 1997).

The soil can usually recover from clogging caused by organic materials, through natural decomposition processes during non-application or resting periods (Loehr et al., 1979; USEPA, 1981; Vandevivere and Baveye, 1992). Balks (1995) has shown the recovery of soil infiltration to be temperature dependent – being slower at $13^{O}C$ than at $25^{O}C$. It was estimated the recovery of soil infiltration in effluent treated soils may occur in less than three week in summer, but may take twice as long in winter.

2.4.4 Soil Strength

Accumulation of soil organic matter and an increase in soil moisture retention may also lead to a decrease in soil strength, measured by a decrease in penetration resistance. Stevens *et al.* (1988) found that applications of pig slurry at rates greater than 250 m^3 ha^{-1} $year^{-1}$ for a total of eight years caused a significant decrease in soil penetration resistance at a depth of 10 to 15 cm.

2.4.5 *Aggregate stability*

The main agents of soil aggregate stabilisation are organic materials, including the products of decomposition of plant and animal remains, the microorganisms themselves, and the products of microbial synthesis (Balks, 1995).

Soil aggregate stability has been shown to increase as a result of additions of organic materials such as animal manures, compost and sewage sludge (Klute and Jacob, 1949; Weil and Kroontje, 1979; Pagliai *et al.*, 1981; Lynch and Bragg, 1985; Kinsbursky *et al.*, 1989). The effect of the added organic material depends on the texture of the soil - sandy soils with low stability respond more than clay soils with inherently high stability (Mbagwu and Piccolo, 1990). For example, additions of pig slurry, sewage sludge and cattle slurry to a sandy loam soil were reported to increase soil aggregate stability by 34, 41 and 26%, respectively (Mbagwu and Piccolo, 1990), whilst additions to a clay soil had no significant effect.

Improved soil aggregate stability may lead to increased macropore space, aeration and root growth. This may help stabilise soils against erosion (Loehr *et al.*, 1979; Khaleel *et al.* 1981) and some investigators have reported decreased runoff volumes from waste-incorporated plots (Khaleel *et al.*, 1981).

Not all wastes will improve soil aggregate stability. Lieffering and McLay (1996) have shown that soil aggregate stability can be significantly reduced by the application of hydroxide solutions, typically present in high pH industrial wastes. This reduction in aggregate stability was attributed to the dissolution of organic matter by the high pH solution, which resulted in a loss of inter-particle bonding. Results from the Wagga Wagga effluent plantation project showed that an increase in exchangeable sodium percentage from 2 to 25% after 5 seasons of irrigation with sewage effluent caused an increase in the tendency of the soil to disperse (Balks *et al.*, 1996).

2.5 Effect of effluent on soil chemical properties

There is a large body of literature on the effects of effluent on soil chemical properties. This section briefly summarises some of these effects, particularly those reported in New Zealand.

The dairy industry and regulatory authorities are interested in the assimilation of applied nutrients within the soil, specifically nitrogen and phosphorus, which have been linked to eutrophication of surface waters. Soil chemical and biological properties have an important role in the assimilation of nutrients from effluent (Balks, 1995).

Many studies have shown that soil chemical fertility is increased after land application of wastes (e.g. Keeley and Quin, 1979; Hart and Speir, 1992). A study of the effects of over 80 years of meatworks effluent to Lismore stony silt loam at Fairton in Canterbury, has shown that considerable increases in soil fertility can occur (Table 2.2) (Keeley and Quin, 1979).

Table 2.2 Nutrient status of Lismore soil (0-15 cm) after 80 years of meatworks effluent application (Keeley and Quin, 1979)

Soil Property	Treatment Area	Control	Soil Property	Treatment Area	Control
pH	6.4	6.1	Extract. SO_4-S ($\mu g\ g^{-1}$)	27	3
Organic Matter			Cations ($cmol_c\ kg^{-1}$)		
C (%)	4.45	3.88	Ca	15.4	9.4
N (%)	0.46	0.36	Mg	1.0	1.0
C/N	9.7	10.8	Na	1.2	0.5
Total P ($\mu g\ g^{-1}$)	1500	630	K	1.4	0.5
Available P ($\mu g\ g^{-1}$)	270	30	Base saturation (%)	87	66

The effects of effluent irrigation on the following soil properties are discussed in more detail: soil pH, carbon, nitrogen, cation exchange capacity, exchangeable sodium percentage and phosphorus.

2.5.1 Soil pH

Sodium hydroxide is often used as a cleaning agent to remove fat deposits in many industrial and food processing plants. The wastes from these plants are alkaline, with pH values often greater than 10 (Lieffering and McLay, 1995). Land application of these alkaline wastes can therefore increase soil pH (Kardos and Sopper, 1973; Keeley and Quin, 1979; Campbell *et al.*, 1980; Ross *et al.*, 1982; Speir *et al.*, 1987; Kannan and Oblisami, 1990; Schipper *et al.*, 1996; Falkiner and Smith, 1997). Keeley and Quin (1979) and Barnett and Parkin (1985) have reported significant increases in soil pH following the application of fellmongery effluent and dairy factory effluent, respectively. Campbell *et al.* (1980) reported increases in soil pH following the application of woolscour effluent to a Waimakariri sandy loam in Canterbury. Falkiner and Smith (1997) also found increases in the pH of forest soils following the irrigation of secondary-treated sewage effluent.

The change in soil pH due to effluent irrigation depends on a number of factors that include the pH of the effluent, the pH buffering capacity of the soil and the effluent loading rate. Not all alkaline effluents have been shown to increase soil pH. At the Fairton meat processing plant, Keeley and Quin (1979) found that there was no change in soil pH despite 80 years of application of an effluent with a pH value of 10. Similar effects have been reported for soils irrigated with dairy factory effluent (Wells and Whitton, 1966) and for irrigation of whey (McAuliffe, 1978).

There is little or no literature available which shows a decrease in soil pH after the irrigation of effluent.

2.5.2 Carbon

Soil organic matter performs a number of functions in the soil, which contribute greatly to general soil fertility. Organic matter storage is recognised as an important indicator of soil quality (Doran and Parkin, 1994; Carter *et al.*, 1997).

The soil microbial population has a large capacity to degrade organic matter added in wastes. Initial rates of degradation are generally fast as there is a flush of biological activity. The microbial biomass then decreases as the easily degradable materials and the microbial metabolites are mineralised (Boyle and Paul, 1989).

Although most effluents contain soluble carbon, the annual rate of organic matter input in effluent treatment systems is generally regarded as being too low to cause a significant increase in soil organic carbon content (Cameron *et al.*, 1997). However, there are many reports where the application of solid wastes and sludges has been shown to cause a significant increase in soil organic carbon (Sopper, 1992). Stadelmann and Furrer (1985) reported that the organic carbon content of a sandy loam soil increased from 1.5% to 2.6% after 7 years of applying sewage sludge at a rate of 5 t ha^{-1} yr^{-1}. Similarly, 10 years of biannual applications of dry cattle manure at 40 and 80 t ha^{-1} were reported to increase the C, N and carbohydrate contents of a silt loam soil in Canada (Angers and N'Dayegamiye, 1991).

McAuliffe (1978) measured organic carbon at two dairy factory land treatment sites in the central North Island, both located on silt loam soils. At the Te Rehunga site, organic C, as measured down the profile to 60 cm, was significantly greater on soils irrigated with effluent compared with control soils as shown in Table 2.3.

Table 2.3 Organic C levels for treatment and control sites at Te Rehunga and Tokomaru dairy factory land treatment sites (McAuliffe, 1978)

Depth (cm)	Te Rehunga Site		Tokomaru Site	
	Treatment	Control	Treatment	Control
0-10	11.25	8.19	5.08	4.69
10-20	7.58	4.33	3.48	2.84
20-30	6.01	2.66	1.97	1.35
30-40	3.60	1.47	1.09	0.95
40-50	2.97	1.05	0.70	0.71
50-60	2.28	0.71	0.56	0.57

Application of untreated meat processing effluent since 1899 has been reported to cause an increase in soil organic C from 5.6 to 6.8%, and increases in microbial biomass, soil respiration, mineralisable N, available N, available P and enzyme activities (Ross *et al.*, 1982). Although this may be seen as beneficial to soil fertility and plant growth, it can also result in substantial amounts of nutrients being leached from the soil (Keeley and Quin, 1979).

Not all wastes lead to an increase in soil organic matter. Land application of industrial wastes with a high pH (10-13) is likely to dissolve soil organic carbon (Lieffering and McLay, 1996), which may in turn lead to the development of soil physical problems.

2.5.3 Nitrogen

Most organic wastes are rich in nitrogen and the land application of these wastes can produce significant increases in soil nitrogen (Wells and Whitton, 1970). Mbagwu and Piccolo (1990) reported that applications of pig slurry (40 t ha^{-1}), cattle slurry (8 t ha^{-1}) and sewage sludge (200 t ha^{-1}) increased the total soil nitrogen content by 18, 13, and 57%, respectively.

Eighty years of land application of meatworks effluent at Fairton increased total nitrogen at the two soil depths sampled. At 0-15 cm, total nitrogen of irrigated soil increased to 0.46%, compared with 0.35% for non-irrigated soils. At 15-30 cm, total nitrogen of irrigated soil increased to 0.36% compared with 0.25% for non-irrigated soils (Keeley and Quin, 1979).

McAuliffe (1978) measured total nitrogen to a depth of 60 cm (10 cm increments), at two dairy factory land treatment sites, both located on silt loam soils. At the Te Rehunga site, total nitrogen was significantly greater at all sample depths on irrigated soils compared with control soils. The greatest increase in total nitrogen was reported at 0-10 cm depth, 1.15% total nitrogen for irrigated soil compared with 0.55% for control soils. At the Tokomaru site, increases in total nitrogen were observed to a depth of 40 cm only. The greatest increase in total nitrogen was reported at 20-30 cm depth, 0.27% total nitrogen for irrigated soils compared with 0.17% for control soils.

Degens *et al.* (2000) reported an increase of 2.1 t ha^{-1} of total nitrogen in the 0-7.5 cm layer on Horotiu silt loam soils in the Waikato, irrigated with dairy factory effluent for 22 years.

In most studies the effect of effluent irrigation on mineral nitrogen has been examined. This gives an indication of plant available N and mobile N (Magesan *et al.*, 1998).

2.5.4 Cation Exchange Capacity

The cation exchange capacity (CEC) of a soil is a quantitative measure of the soil's ability to hold exchangeable cations. The proportion of the exchange capacity that is occupied by cations is referred to as the percentage base saturation (%BS).

The total CEC and BS are important as they determine a soil's capacity to remove dissolved inorganic cations and other positively charged organic ions from effluent. Cations, including heavy metals accumulate in soil until the soil cation exchange complex becomes saturated.

An increase in soil organic matter content resulting from the application of organic wastes, can produce a related increase in soil cation exchange capacity. Bernal *et al.* (1992) reported that the application of pig slurry at a range of rates from 200 to 1000 m^3 ha^{-1} yr^{-1} resulted in significant increases in soil organic carbon and cation exchange capacity. Similarly, Stadelmann and Furrer (1985) reported that 7 years of application of sewage sludge and pig slurry at 5 t ha^{-1} to a sandy loam soil increased the CEC from 17.2 $cmol_c$ kg^{-1} in the control plots, to 23.7 and 22.2 $cmol_c$ kg^{-1} in the treated plots, respectively.

Where effluent also contains significant quantities of the exchangeable basic cations Ca^{2+}, Mg^{2+}, Na^+, and K^+, the base saturation of the soil may also be increased. Keeley and Quin (1979) reported a significant increase in base saturation with application of fellmongery effluent (Table 2.2). Campbell *et al.* (1980) reported a significant increase in base saturation (from 78% in control soils to 103% in irrigated soils at 0-15 cm depth and from 75% in control soils to 102% in irrigated soils at 15-30 cm depth) and exchangeable K^+ following application of wool scour effluent for 7 years.

2.5.5 Exchangeable sodium percentage

A high concentration of sodium in soil is of concern because it can cause a reduction in soil aggregate stability that promotes surface sealing, pugging and grass pulling. This can cause a decrease in infiltration rate and an increased risk of surface runoff. High salt concentrations in soil solution can also reduce the soil osmotic water potential and thus decrease the amount of water that is readily available for plant uptake (Cameron *et al.*, 1997).

Sodicity can develop when effluent with a high sodium concentration is applied (Sumner and McLaughlan, 1996). The application of secondary-treated dilute sewage effluent to a soil at Woolgoola, NSW was reported to more than double the soil sodium concentration from 0.11 to 0.31 cmol (+) kg^{-1} (Johns and McConchie, 1994). Despite the low electrical conductivity of the effluent (0.44 dS m^{-1}) the soil exchangeable sodium percentage (ESP) values reached 4% during the trial.

The application of effluent from a pulp and paper mill in Kawerau, New Zealand was reported to have caused an increase in sodium adsorption ratio (SAR) from 2 to 16 and an increase in sodium concentration in groundwater (Johnson and Ryder, 1988).

2.5.6 Phosphorus

Organic wastes are often valuable sources of P, which may be used to increase soil P status (Wells and Whitton, 1966; Wells and Whitton, 1970; Quin and Woods, 1978). Mbagwu and Piccolo (1990) reported that applications of pig slurry (40 t ha^{-1}), cattle slurry (8 t ha^{-1}) and sewage sludge (200 t ha^{-1}) increased Olsen P by 430, 372 and 642%, respectively. Keeley and Quin (1979) reported an increase of total and Olsen P after the addition of meatworks effluent for eighty years, at two soil depths sampled, as shown in Table 2.4.

Table 2.4 Olsen P and total P levels for treatment and control sites at the Fairton meatworks land treatment site (Keeley and Quin, 1979)

Depth (cm)	Olsen P (ug g^{-1} soil)		Total P (ppm)	
	Treatment	Control	Treatment	Control
0-15	630	30	1 500	630
15-30	520	15	1 200	520

McAuliffe (1978) measured Olsen P at two dairy factory land treatment sites. At both the Te Rehunga and Tokomaru sites, Olsen P measured down the profile to 20 cm and 30 cm, respectively, was significantly greater on irrigated soils compared with control soils, as shown in Table 2.5.

Table 2.5 Olsen P levels for treatment and control sites at Te Rehunga and Tokomaru dairy factory land treatment sites (McAuliffe, 1978)

Depth (cm)	Te Rehunga Site		Tokomaru Site	
	Treatment	Control	Treatment	Control
0-10	660	76	252	55
10-20	320	24	278	30
20-30	-	-	151	19

Degens *et al.* (2000) reported an increase of 11.5 t ha^{-1} of total P at 0-75 cm depth on Horotiu silt loam soils, irrigated with dairy factory effluent for 22 years. Similar patterns occurred in the Olsen P levels, but the magnitude of the difference was much greater, up to 22 fold greater on irrigated soils at 0-25 cm depth. This indicated that some of the P storage occurred in available forms that may benefit future pasture production. Continued accumulation of P in soils, however may result in P levels toxic to plants or induce deficiencies of other nutrients in the pasture (Marschner, 1986). Pasture levels of P at the Waikato site ranged between 0.50 and 0.75% and no toxicity effects have been observed (Degens *et al.*, 2000).

2.5.7 Phosphate Retention

The removal of phosphorus from effluent by soil is achieved through the processes of precipitation or adsorption by soil colloids. Removal is dependent on factors such as soil texture, soil pH, presence of calcium, and the amount of iron and aluminium oxides or aluminosilicates (e.g. allophane) present. Although most soils have a large capacity for phosphorus removal from effluent, phosphorus sorption capacity is finite and over a long period of time, or with heavy loadings, the soils' ability to remove phosphorus from applied effluent may be exceeded (USEPA, 1976; 1981). The time period over which a soil will continue to intercept phosphorus without leaching to groundwater is generally unknown (USEPA, 1981).

Phosphorus retention mechanisms in most New Zealand soils are considered to result in a low risk of P leaching (White and Sharpley, 1996). At the 'Flushing Meadows' scheme at Wagga Wagga, New South Wales, the depth of P leaching was reported to be less than 2.5 cm after 2½ years of application of treated sewage effluent (Falkiner and Polglase, 1996). However, phosphorus leaching is a concern in some sandy soils.

The potential for phosphorus leaching depends on the P levels in the soil. Soils from the Broadbalk experiment at Rothamsted, which have received annually varying amounts and forms of P for more than 100 years, now contain a wide range of Olsen P concentrations in the plough layer. It was found up to 60 mg Olsen P kg^{-1} soil, P was retained strongly in the plough layer. Above this P value losses in the drainage water were much more closely related to Olsen-P than commonly suggested (Heckrath *et al.*, 1995).

2.6 Effect of effluent on soil biological properties

In most pasture soils, the micro-organism and earthworm populations make up the bulk of the soil biomass (McLaren and Cameron, 1996). Both of these groups are of importance with regard to land treatment of effluent. Biological processes centre around soil micro-organisms, which alter waste constituents through decomposition, inorganic and organic transformations and nutrient assimilations. Biological processes are generally restricted to approximately the upper metre of soil (Loehr *et al.*, 1979) and mainly occur in the topsoil where the highest organic matter contents are found (Balks, 1995).

2.6.1 Earthworm populations

Earthworms have been identified as potential indicators of soil health and environmental quality (Beare *et al.*, 1997). They are a useful gauge of the condition of the soil at a land treatment system. Earthworms are beneficial at an effluent treatment site, as they contribute to the breakdown of the effluent, as well as assisting in maintaining (or improving) soil permeability, aeration and soil structure development (Balks, 1995). Earthworm burrowing activity is important to maintain soil permeability and in many treatment systems it is essential to maintain adequate soil hydraulic conductivity (Jones *et al.*, 1993).

Organic wastes provide organic matter that can act as a carbon source and thereby lead to an increase in earthworm numbers (Neuhauser *et al.*, 1980; Edwards and Lofty, 1982). The greater microbial biomass in effluent irrigated soil, can also contribute to greater survival and activity of earthworms (Ross *et al.*, 1982; Cameron *et al.*. 1997).

Applications of effluent can alleviate seasonal moisture limitations in soils, thereby enabling longer periods of earthworm activity in irrigated soils compared with non-irrigated soils (Baker. 1998). Yeates (1995) reported earthworm populations responded significantly to 7 years of sewage effluent application to *Pinus radiata* at Waitarere, New Zealand. An increase in soil moisture was said to be the key factor in this response to effluent application. A study at Massey University assessed the effect of dairy effluent treatment on earthworm populations (Yeates. 1976). Earthworm numbers and weight were doubled on the irrigated area. The difference in population was attributed to the different rates of dry matter production and reduced earthworm mortality, due to maintenance of high soil moisture, coupled with increasing soil temperatures.

McAuliffe (1978) reported that earthworm numbers at the Te Rehunga and Tokomaru dairy factory land treatment sites, were five to six-fold higher than the corresponding control sites. There was no change in earthworm populations reported at the Longburn site. This was attributed to shorter time of effluent application and spatial variation in earthworm numbers. Degens *et al.* (2000) reported non-irrigated allophanic soils had more earthworms than soils irrigated with dairy factory effluent. However, the irrigated site had a greater biomass of earthworms (Table 2.6) and the mean individual biomass of earthworm was greater under irrigated than non-irrigated pasture (0.85 g vs 0.51 g). At the control site, earthworms were found down to 40-50 cm depth, with 61% of the biomass being in the 0-20 cm layer. In contrast, under irrigation all earthworms were found above 40 cm, and the 0-20 cm depth contained 90% of the earthworm biomass.

Table 2.6 Vertical distribution of earthworms at effluent and control sites (mean ± standard error) (Degens *et al.*, 2000)

Depth (cm)	Abundance (individual m^{-2})		Biomass ($g\ m^{-2}$)		Soil Moisture (% oven dry soil)	
	Control	Irrigated	Control	Irrigated	Control	Irrigated
0-10	172 ± 38	139 ±23	43 ±12	68 ±6	53.9	75.0
10-20	90 ±13	63 ±2	52 ±7	96 ±18	55.1	65.2
20-30	22 ±4	9 ±3	33 ±7	12 ±4	61.2	62.8
30-40	14 ±4	3 ±2	19 ±5	5 ±3	59.8	63.6
40-50	5 ±3	0 ±0	8 ±5	0 ±0	60.7	-
Total	303 ±51	214 ±33	155 ±20	303 ±51	-	-

Earthworms can respond rapidly to changes in farm management practices (Lee, 1985; Edwards and Bohlen, 1996; Fraser, 1994; Fraser *et al.*, 1996). There are a number of factors that can influence the health of earthworm populations at a land treatment system. Earthworms cannot survive extended periods of soil saturation, so if effluent application saturates the soil, it may be detrimental to earthworm populations. Soil compaction and soil pugging caused by grazing stock, may also reduce the abundance of earthworms.

2.6.2 Respiration rates

The soil microbial population is a mixture of bacteria, actinomycetes, fungi, algae, and protozoa. Soil respiration is often used as an index of microbial activity. The rate of biodegradation of applied organic material depends on the nature of the organic components, the amount of solids in the effluent, the amount and frequency of effluent applications (Loehr *et al.*, 1979; Fuller and Warrick, 1985) and the microbial activity. When organic material is added to soil, firstly the organisms digest the more easily decomposed material, consuming oxygen and releasing carbon dioxide in the process. This pulse of activity can result in a rapid increase in organism numbers/and synthesis of new product such as microbial protein. The organism numbers decline as the easily decomposed fraction is exhausted and reach about the same level as prior to effluent addition (Loehr *et al.*, 1979).

In natural systems that receive localised or periodic additions of organic matter, there is usually a rapid increase in soil respiration rates following effluent application. For example, simulated whey effluent application to soil induced a marked increase in the respiration rate within a few hours of application (McAuliffe, 1978). However, a slower response was recorded by Jewell and Loehr (1975) who showed that oxygen uptake following application of food processing effluent to soil peaked within two days, but did not return to initial levels after 5 days. They also showed that the oxygen uptake increased with the rate of effluent application.

The soil microbial community changes in response to effluent application. Soils that have previously received effluent have been observed to have a quicker increase in respiration rate, following whey addition (McAuliffe, 1978) and a greater total oxygen uptake following food processor effluent application (Jewell and Loehr, 1975), than non-irrigated soils.

Degens et al. (2000) reported basal respiration rates were substantially greater ($P \leq 0.001$) under soils irrigated with dairy effluent (irrigated: 6.24 μg C g^{-1} soil h^{-1}; non-irrigated: 4.02 μg C g^{-1} soil h^{-1}). Substrate use by microbial communities in the irrigated soil also differed from those in the non-irrigated soil. Mineralisation of organic substrates in the irrigated soil was greater than in the non-irrigated soil. The ratio of substrate use in irrigated to non-irrigated soil revealed that, for most substrates, substrate use was ($P \leq 0.05$) increased to between 1.5 and 3.0 fold greater in the irrigated soil. For lactose and glucose, however, substrate use was 5.6 and 9 fold, respectively, greater in the irrigated soil.

Although most studies report an increase in soil microbial activity following effluent addition, this is not always the case. In a study where soil cores were irrigated with treated sewage effluent or tap water, differences between soils were found to be greater than differences between treatments, for total microbial populations and respiratory activity (Cairns *et al.*, 1978).

2.7 Effect of effluent on water quality

When organic waste material is applied to soil in excessive amounts or under adverse conditions, it can cause contamination of lakes, streams, rivers and underground aquifers. At the majority of land treatment sites in New Zealand, both surface and groundwater may be susceptible to contamination. Groundwater is susceptible to leaching of nutrients through the soil profile and surface water is at risk from surface runoff of effluent and/or rainfall containing sediment and nutrients.

2.7.1 Effect of effluent on surface water quality

Application of effluent at rates exceeding the infiltration rate of the soil and/or application of effluent to soils already saturated are the two main causes of surface runoff at land treatment sites. Nutrient enrichment of surface water (eutrophication), can result in increased growth of algae and other higher aquatic plants (McLaren and Cameron, 1996). This is usually unsightly and also creates problems for recreational activities. When these aquatic plants and algal blooms die, their decomposition depletes the water of dissolved oxygen, a process that also occurs when organic substances, such as sewage or farm effluents, are discharged into water bodies (McLaren and Cameron, 1996).

Surface runoff can be mitigated by a number of measures including: selection of an area to be irrigated that is least susceptible to surface runoff in terms of moisture content, slope, distance to surface water, timing the effluent application to avoid heavy rainfall and planting of riparian zones to intercept any runoff that does occur.

2.7.2 Effect of effluent on groundwater quality

Groundwater quality is an issue at many land treatment sites, especially if groundwater is used as a potable water supply. The major concern with groundwater quality is the leaching of nutrients, particularly nitrogen. Nitrate leaching from agricultural land has been recognised as a major cause of groundwater contamination in many countries (Barraclough *et al.*, 1992; Jarvis, 1993; Spalding and Exner, 1993; Addiscott, 1996; Cameron *et al.*, 1997). While appropriate concentrations of mineral nitrogen in the soil are necessary for plant growth, large nitrogen inputs in the form of organic wastes can increase nitrate leaching (McLaren and Cameron, 1996).

Because high concentrations of nitrate in drinking water are detrimental to human health, the World Health Organisation (WHO) has set a guideline for drinking water of 11.3 mg NO_3^-N L^{-1}. (Cameron et al., 1996). Regional authorities in New Zealand have also established guidelines on land application of wastes and in some cases fertilisers to minimise contamination of water resources by farming activities (Cameron et al., 1996).

Nitrate leaching is affected by many soil, environmental and management conditions (Garwood and Ryden, 1986). Organic wastes differ significantly in their properties and composition. Therefore, their potential for causing nitrate leaching may vary considerably and be different from that of chemical fertilisers (Steenvoorden *et al.*, 1986; Cameron *et al.*, 1996; 1997). Consequently, every land treatment system will have a different impact on groundwater quality. For these reasons, it is difficult to compare results from different leaching experiments, because of the differences in effluent composition, soil type, climatic conditions and management practices.

The rate of waste application has a considerable influence on nitrogen leaching losses. Excessive application rates of nitrogen can pose a significant threat to water quality. Keeley and Quin (1979) observed that at the Fairton meat-processing plant, effluent irrigation nutrient loadings have been in excess of plant requirements. Most of the applied nutrients, with the exception of phosphorus are lost in drainage water and eventually reach groundwater. Keeley and Quin (1979) calculated that greater than 40% of applied nitrogen was leached below 40 cm depth in Lismore stony silt loam pasture soils, which received meatworks effluent at a rate of 900 kg N ha^{-1} $year^{-1}$. The groundwater at the plant has a similar chemical composition to the effluent except that nitrate-N was higher and some N was not accounted for (Keeley and Quin, 1979). Lysimeter studies have confirmed that nitrate leaching losses are small when wastes are applied at relatively low rates but that leaching losses are considerably greater at higher rates of application (Cameron *et al.*, 1995). Results from the Wagga Wagga effluent plantation project (Polglase *et al.*, 1994) also show that irrigation of treated sewage effluent at twice the rate of water use by gum trees (*Eucalyptus grandis*) and pine trees (*Pinus radiata*) can result in increased nitrate leaching losses (Bond *et al.*, 1996). Application of effluent at a rate that matched plant water use did not result in an adverse impact on groundwater quality (Bond *et al.*, 1995).

In a trial where meat processing effluent was applied to an allophanic soil in the Waikato, Russell *et al.* (1984) found that the major nutrient lost in the water percolating from the soil was nitrate-nitrogen. Typical annual nitrate nitrogen losses from the control and irrigated plots are shown in Table 2.7.

Table 2.7 Nitrate-nitrogen losses from control and soil irrigated with primary treated meat-processing effluent

Annual effluent loading (kg N ha^{-1})	Annual amount NO_3-N leached (kg N ha^{-1})	NO_3-N concentration (g m^{-3})
0	10	1.3
600	100	9.6
1200	330	23.0
2600	1700	70.0

Losses of nitrate to groundwater increased as the amount of N applied was increased. Irrigation with primary treated meat-processing effluent at loadings up to 1200 kg N $ha^{-1}y^{-1}$ produced groundwater effects similar to alternative land uses such as pastoral farming.

2.8 Effect of effluent on pasture

The objective of land treatment of wastes is to utilise the chemical, physical and biological properties of the soil/plant system to assimilate the waste components without adversely affecting soil quality or causing contaminants to be released into water or the atmosphere (Loehr, 1984). Pasture growth on a land-treatment site is an essential feature of the system, as it removes nutrients and water from the waste treated soil, as well as protecting the soil from damage during waste application (Cameron *et al.*, 1997). Therefore it is important that effluent irrigation causes minimum detrimental effect on both the quantity and quality of pasture.

Irrigation of effluent often results in increased dry matter yield (Hart and Speir, 1992; Cameron *et al.*, 1997; Edwards and Daniel, 1992; Cameron *et al.*, 1996). Increased pasture production has been reported for sites under dairy factory effluent irrigation and attributed to the high levels of nitrogen and potassium added in the effluent (Wells and Whitton, 1970). Increases in pasture production from 81 to 174 % have been recorded at New Zealand sites receiving regular applications of meat processing plant effluent, throughout the growing season (Russell and Cooper, 1987).

Land application of dairy shed effluent has been shown to be very effective in stimulating pasture growth (MacGregor *et al.*, 1979; Goold, 1980; Cameron *et al.*, 1996). Yeates (1976) reported that pasture growth was increased from 12 000 to 16 000 kg DM ha^{-1} yr^{-1} with the application of dairy effluent. Goold (1980) reported a 27% increase in pasture production with 6 mm of dairy shed effluent applied every 21 days (representing 156 kg N, 46 kg P, and 348 kg K ha^{-1} yr^{-1}) and a 43% increase with 12 mm application to a clay soil in Northland. The effluent effects are particularly significant in spring, summer and autumn.

Land application of sewage effluent has also been reported to cause significant pasture yield increases. Quin and Woods (1978) showed that pasture response to the applied nutrients (equivalent to 116 kg N, 34 kg P and 68 kg S ha^{-1} yr^{-1}) was greatest in summer and autumn. Cameron *et al.* (1995) measured a 70% increase in pasture production from both a low and a high rate of application of pig slurry (200 and 600 kg N ha^{-1} $year^{-1}$).

There is also the potential for effluent application to have detrimental effects on pasture growth. Pasture "burning" (i.e. dieback due to effluent contact with the foliage), as a result of whey application was observed at all sites investigated by McAuliffe (1978) and has been reported elsewhere by Parkin and Marshall (1976). Pasture burning not only results in decreased dry matter production, but can also encourage infestation of weeds in scorched areas. This has been attributed to various factors such as: insufficient dilution, effluent with a high pH, effluent that has gone septic as a result of prolonged retention in lines or holding tanks, effluent that was too hot and spraying onto pasture that was too long.

'Pasture pulling' (where grazing animals uproot plants) is a regular problem at grazed land treatment sites. Thomas and Evans (1975) noted that pasture pulling generally occurred on free draining soil, particularly after heavy rain following a dry spell. Pasture pulling was reported as being more extensive in shallow-rooted plant species, particularly if they were growing in a highly fertile soil. It was also noted that 'pasture pulling' appeared to be more common in late spring and autumn and over the summer period, when plants were of a more stalky nature. Pasture pulling at whey application sites was reported by McAuliffe (1978). He suggested that this was probably the result of loss of soil strength, due to high moisture content and a tendency for plants to only develop shallow root systems under effluent irrigation.

The policy of effluent application to pasture is sometimes made without due regard to its impact on the quality of pasture as an animal feed. For example, the supply of large quantities of selective nutrients, such as nitrogen and potassium, through effluent irrigation could affect the nutrient balances and the botanical composition of the pasture (Campbell *et al.*, 1980). It has also been claimed that excessive effluent irrigation leads to poor utilisation of pasture and nutrient-related disorders in grazing animals (Bolan *et al.*, 2000).

A field experiment was conducted at Massey University No. 4 Dairy Unit to examine the effect of effluent application rate on the nutrient status of pasture (Bolan *et al.*, 2000). Three levels of effluent irrigation (at nitrogen loading of 0, 150 and 200 kg ha^{-1}) were applied. Effluent irrigation at these three rates also provided 0, 256 and 342 kg ha^{-1} of potassium. The effluent was irrigated both in the presence and absence of calcium (gypsum) and magnesium (Epsom salts) fertiliser additions to pasture. The concentrations of various nutrient anions and cations in the pasture were measured. The dietary cation-anion difference (DCAD) values and grass stagger index (GSI) were also calculated.

$$DCAD = ([Na^+] + [K^+]) - ([Cl^-] + [SO_4^{2-}])$$

$$GSI = [K^+] / ([Ca^{2+}] + [Mg^{2+}])$$

where: [ ] = milliequivalents/kg-DM

The results from the trial showed effluent irrigation decreased the supply of both Ca and Mg for animal uptake. The supply of Ca was higher than the maintenance requirements of dairy cows. However at the highest level of effluent irrigation the supply of Ca was less than the lactation and pregnancy requirements. The supply of Mg was higher than maintenance, lactation and pregnancy requirements by dairy cows even at the highest level of effluent application. It is probable that potassium loading through the effluent irrigation decreased the uptake of Ca and Mg by pasture. This is mainly due to the leaching of these cations resulting from the competition of K^+ ions and the supply of mobile companion nitrate and chloride ions (Rhue and Mansell, 1998). A high concentration of potassium in soil solution is likely to result in luxury uptake of potassium by plants. This will lead to less uptake of other cations in order to maintain cation-anion charge balance in plants (McNaught, 1959).

The dietary cation anion difference (DCAD) ranged from 220 – 740 meq kg^{-1} DM and it increased with increasing levels of effluent irrigation. It has been observed that DCAD values for most pastures in New Zealand range from 200 to 800 meq kg^{-1} DM, indicating cation surplus for anions (Bolan *et al.*, 2000). This normally results from the luxury uptake of potassium by pasture resulting in high concentration of potassium in herbage. Dietary cation-anion balance is important because animals attempt to maintain systemic acid-base balance and osmotic pressure in order to protect the integrity of cells and membranes and to optimise biochemical and physiological processes (Wilson, 1999). When herbage with excess cations over anions (positive DCAD) is fed to animals the concentration of alkali ions such as bicarbonate ions increases in body fluids resulting in alkalosis. Conversely, if feed with surplus of anions (negative DCAD) is ingested then the concentration of acidic hydrogen ions increases and metabolic acidosis occurs. There have been conflicting reports on the optimum levels of DCAD values required for dairy cattle (Roche, 1997). It has however been shown that diets with high DCAD values tend to increase the incidence of milk fever and supplementation of pre-calving rations with anionic salts (low DCAD values) reduce the incidence of milk fever (Beede, 1992; Wilson, 1996).

The GSI values measured in the trial ranged from 1.35–3.46 and increased with increasing levels of effluent irrigation. Application of gypsum and Epsom salts resulted in a decrease in DCAD and GSI values, mainly due to an increase in the concentration of calcium and magnesium. It has been suggested that when GSI values in the pasture are in excess of 2.2 the risk of hypomagnesaemia (grass staggers) development is enhanced (Mason and Young, 1999). This condition is generally associated with animal serum magnesium levels less than 1.0-1.5 mg 100 ml^{-1}, compared to normal levels in cattle of 1.7-3.0 mg 100 ml^{-1}. This arises due to diets low in magnesium and additionally potassium induced inhibition of magnesium absorption in the rumen (Grunes and Rending, 1979; Roberts, 1994).

CHAPTER 3

Materials and Methods

3.1 Site description

The three farms used in the land treatment system are all farmed as dairy units. The topography of the land is flat to undulating with each farm dissected by at least one or two streams. Pasture species are predominantly ryegrass and clover with some weeds including dock and willow weed. Patches of willow weed occur in some irrigated paddocks. Figures 3.1, 3.2, and 3.3 show site layouts for each individual farm with paddock numbers, locations of streams and the layout of the irrigation system.

3.2 Soils of the land treatment farms

The soil type on the three land treatment farms is the Egmont loam soil, a moderately deep variant, classified as having moderate infiltration. The NZ Classification of Egmont soils is, Typic Orthic Allophanic (Hewitt, 1993). Orthic Allophanic soils were developed inland under podocarp/hardwood indigenous vegetation (Molloy, 1993). The parent material is airfall andesitic ash on medium to coarse textured andesitic alluvium, generally between 3 500 and 50 000 years old (Molloy, 1993). The soils are permeable without barriers to deep penetration of pasture roots. The soils are moderately well, well or imperfectly drained depending on location, position in landscape and texture. The profiles of allophanic soils consist generally of Ah, Bw, and C horizons (Molloy, 1993).

Egmont soils contain a large proportion of the mineral allophane in the clay fraction throughout the soil profile. The C:N:S:P ratio of the topsoils is approximately 100:7.7:1.4:1.2 (McLaren and Cameron, 1996). The soils have very high phosphate retention (generally above 80%), particularly in the subsoils (greater than 90%). The large capacity to adsorb phosphate is one of the soils' most important characteristics (Molloy, 1993) in farming and especially with regard to the treatment of effluent with high P content. Some of the selected chemical characteristics of the Egmont Brown Loam are given in Table 3.1.

Table 3.1 Average values for selected properties in the Egmont soil profile (Wheeler and Edmeades, 1991).

Ca	Mg	Na	K	Al	pH	PR	C	N	Clay
---------------- meq 100 g^{-1} soil ---------------						(%)	(%)	(%)	(%)
10.1	2.20	0.42	0.60	0.38	5.7	81	11.6	0.95	11

On all three irrigation farms there are a couple of subsurface drains through low-lying areas but not widespread artificial drainage. These are mainly to intercept subsurface seepage on low-lying land (e.g at the bottom of gullies etc). There is no regular spacing of drains. Good drainage of Egmont Brown Loams is ensured by the high porosity of the soils (A horizon – 70%, B horizon – 69.5%) (McLaren and Cameron, 1996) which contain an exceptionally high content of large pores. As a result of low bulk density (A horizon – 0.66 gcm^{-3}, B horizon – 0.73 gcm^{-3}) (McLaren and Cameron, 1996) there is little resistance to root extension. Soil biomass is generally high with a high proportion of microorganisms, particularly in the A horizon (Molloy, 1993).

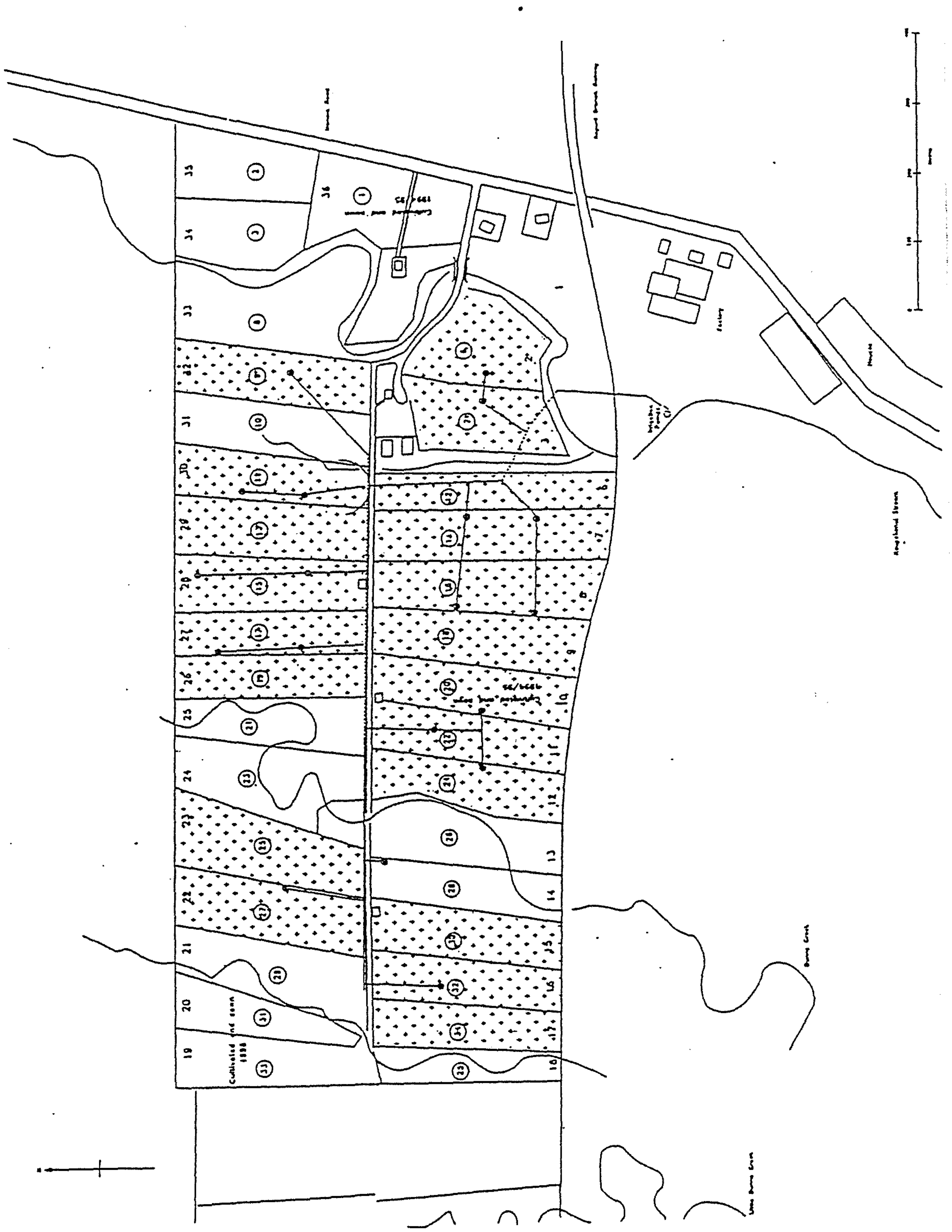


Figure 3.1 *Site Layout of Farm No. 1*

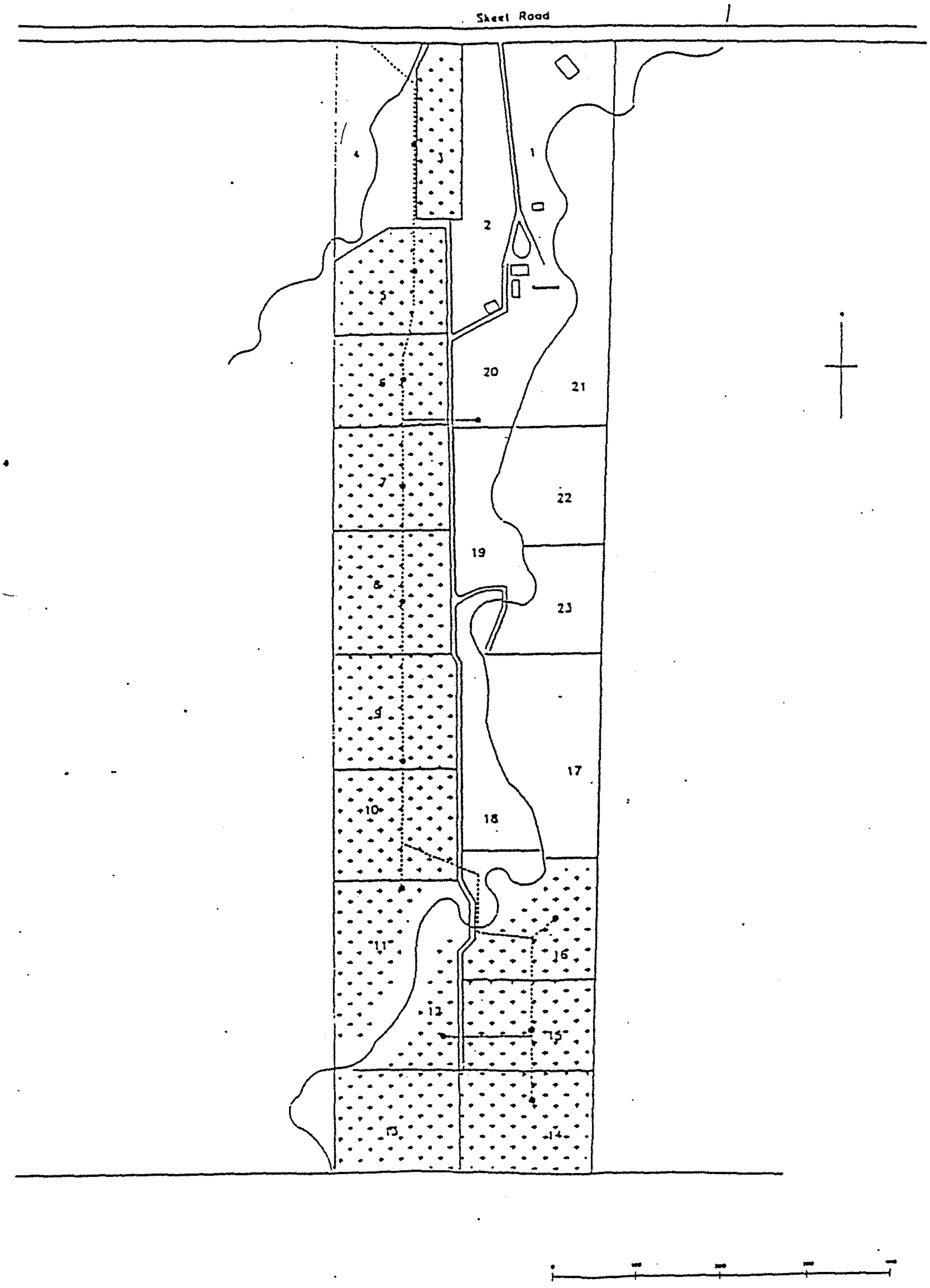


Figure 3.2 *Site Layout of Farm No. 2*

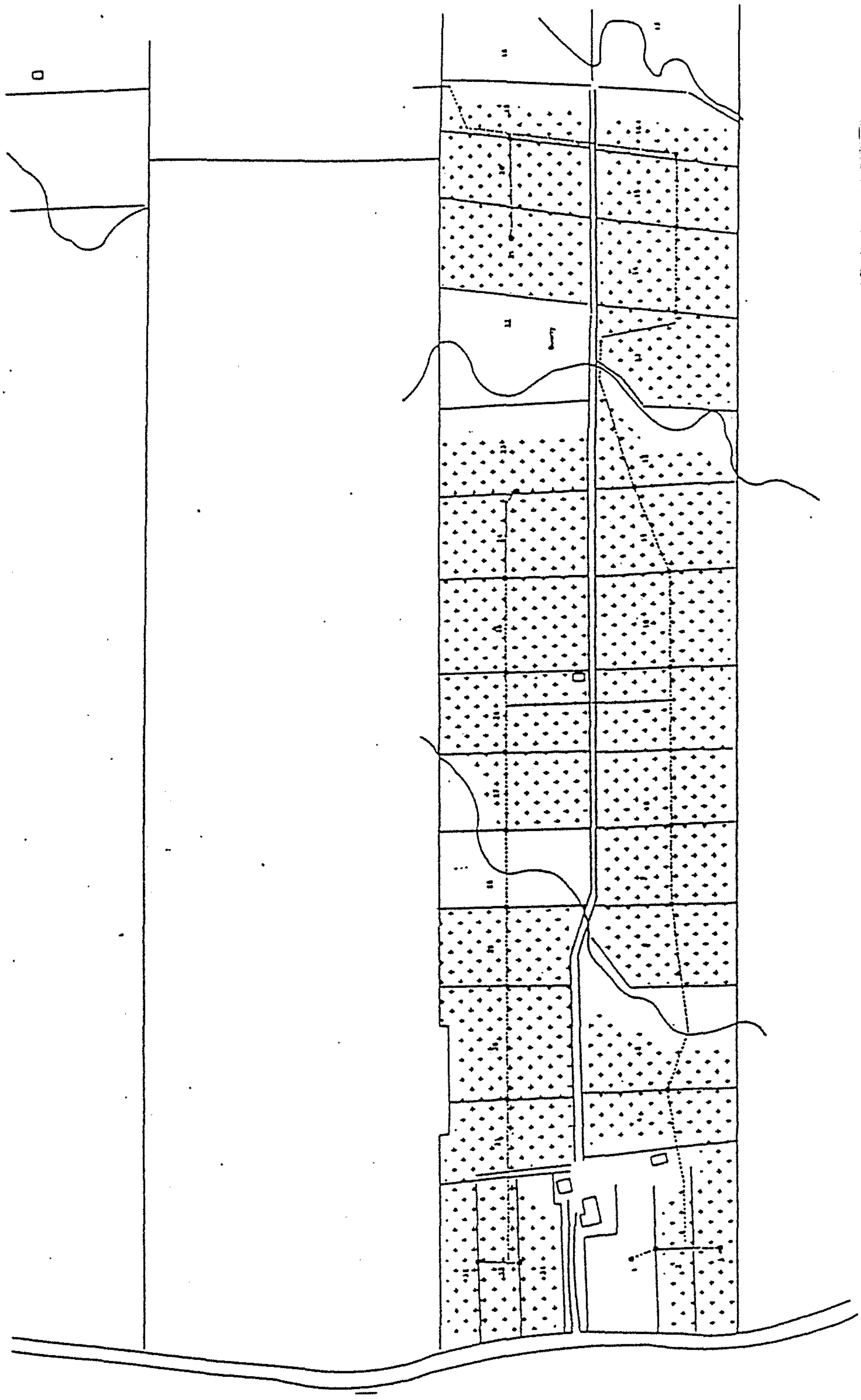


Figure 3.3 *Site Layout of Farm No. 3*

As mentioned in section 1.4, the New Zealand Dairy Research Institute has undertaken soil investigations at the LNZ farms. The following soil profile description shown in Table 3.2, was completed by NZDRI staff in June 1987.

Table 3.2 Soil Profile Description of Egmont Brown Loam (Paddock 24, farm No. 1) (NZDRI, 1988)

0 - 13 cm	Black (10yr 2/1) silt loam. Very friable, moderately developed medium to fine nut structure. Abundant roots. Few earthworms. Distinct worm mixed boundary
13 - 26 cm	Dark brown (7.5yr 3/2) silt loam. Friable. Weakly developed coarse nut breaking to moderately developed medium and fine nut structure. Many roots. Distinct boundary.
26 - 58 cm	Olive brown (2.5yr 4/4) sandy loam. Friable. Weakly developed medium blocky structure. Few roots. Few fine gravels (2-3 mm). Indistinct boundary.
58 - 78 cm	Olive brown (2.5yr 4/4) medium loamy sand. Friable to firm. Massive. Few roots. Indistinct boundary. Few stones up to 8 cm.
78 – 120+ cm	Dark greyish brown (2.5yr 4/2) medium sand. Loose. Single grained. Few fine stones.

On farm No.1, the paddocks on the western side of Dunns Creek have iron concretions in the subsoil. These paddocks tend to have lower surface infiltration rates than the other treatment areas and surface ponding is more apparent. The iron concretions tend to be closer to the soil surface at the 'back' of farm No.1 and paddock 33 is not irrigated due to the proximity of the iron concretions to the soil surface.

3.3 Climate

All quantitative climatic data is from the Manaia Demonstration farm, which is approximately 5 km south of Kapuni. The Kapuni area has a mean annual rainfall of 1224 mm (1950-1980) as shown in Table 3.3. The long term average yearly reference crop evaporation (or potential ET) for Manaia (E94512) is given by the NZ Met Service as 817 mm. Rainfall is evenly distributed with a winter high and summer low. Rainfall for the 1999 year was 1304 mm, 60 mm greater than the mean annual rainfall. Rainfall for the 1999 year was very erratic, characterised by either large (291 mm in November) or very low (4 mm in February) monthly totals. Rainfall data for 1999 was measured in a rain gauge located at farm No.3: daily readings were made by the sharemilker. The average annual air temperature is 12.7^{0}C (1961-1990) and average annual sunshine hours for the area are 1987 hours (1951-1980). Wind is predominantly from the westerly quarter. Average annual soil temperature at 0-10 cm depth is 12.6^{0}C (1951-1980). Monthly values for average air temperature, sunshine hours and soil temperature are shown in Table 3.3.

Table 3.3 Distribution of mean annual and 1999 rainfall, air temperature, sunshine hours, and soil temperature.

Month	Average Rainfall (mm)	1999 Rainfall (mm)	Average air temperature (^{0}C)	Average Sunshine (hours)	Average Soil Temperature (^{0}C)
January	83	165.5	16.6	237	18.1
February	80	4.0	16.7	202	17.9
March	85	93.0	15.9	176	15.9
April	100	82.5	13.5	151	13.0
May	137	49.5	11.3	124	10.3
June	130	117.5	9.5	103	8.1
July	128	191.5	8.6	115	6.9
August	115	140.5	9.2	132	7.7
September	98	71.0	10.5	145	9.6
October	87	36.0	12	180	12.2
November	85	291.0	13.5	198	14.4
December	96	62.0	15.3	224	16.7
Total/Average	1244	1304	12.7	1987	12.6

3.4 Effluent analysis

Three samples of effluent were collected on 21 December 1999 for chemical analysis. The samples were collected from the pump house, which pumps effluent from the balance tank to the irrigators. Samples were collected over a 6 hour period, at the start, middle and end of an irrigation run. The samples were analysed by R J Hill laboratories in Hamilton. Samples were analysed for pH, total suspended solids, total sodium, total potassium, total ammoniacal-N, total Kjeldahl nitrogen (TKN), nitrate-N + nitrite-N (TON), nitrate-N, nitrite-N, total phosphorus, sulphate, carbonaceous biochemical oxygen demand (CBOD~5^) and total chemical oxygen demand (COD).

3.5 Characterisation of the soil resource - measurement of soil properties

The characterisation of the soil resource involves three aspects; measurements of physical, chemical and biological properties. The NZDRI has been monitoring the chemical status of the soil for several years on all three farms. The work has predominantly involved chemical analysis of the surface 0-7.5 cm soil. In addition, there have been very limited measurements of soil physical properties (only one-off measurements of soil moisture contents, bulk density and infiltration rates have been undertaken).

The land treatment system at LNZ has developed considerably since its inception in the late 1950's. As effluent volumes increased and the need to reduce hydraulic loadings emerged, the land area used for effluent irrigation increased several times. Therefore, there are paddocks with varying years of effluent application (Figure 3.4). This allows the effect of effluent irrigation on the soils to be measured at different time intervals over a 40 year period. Table 3.4 shows paddocks used in the characterisation of the soil resource and the number of years these paddocks have been irrigated with effluent.

Table 3.4 The paddocks used in the study and the years these paddocks have received effluent

Years of irrigation	Farm Number	Paddock Number
0	1	5
0	2	2
0	3	17
9	3	10
9	3	24
14	1	25
14	1	32
30	1	17
30	1	18
40	2	10
40	2	16

A total of 11 paddocks were used for the characterisation of the soil resource. For some of the sampling and analysis that is very time consuming, fewer paddocks were used. These were typically, non-irrigated paddocks from each farm, as well as one paddock each from different irrigation periods (paddock 17 farm No.1, paddock 10 farm No. 2, paddock 10 farm No. 3). Although three non-irrigated paddocks were sampled (One from each farm), only two paddocks were sampled per irrigation period. This is not ideal in terms of reducing error associated with variation between paddocks irrigated for the same number of years. However sampling more paddocks to increase the number of repetitions was impractical in terms of cost and time required for sampling and analyses.

3.6 Soil physical measurements

A number of soil physical measurements were undertaken on the study paddocks, including: assessment of soil strength, bulk density, aggregate stability, soil infiltration rates, soil moisture contents, and soil moisture retentivity.

Figure 3.4 *Comparisons of soil profiles with varying years of effluent irrigation*

3.6.1 *Soil moisture contents*

The moisture contents were measured for soil samples collected on six separate occasions: 26-28 April 1999, 23 June 1999, 6-8 July 1999, 10-15 September 1999,29 January 2000 and 23 March 2000. Bulked soil samples were collected from all study paddocks using a soil corer to collect random cores. Samples were placed in plastic bags and stored at 4^{O}C before analysis. The soil samples were dried in an oven at 105^{0}C until further drying produces no loss of weight (for at least 24 hours) to determine volumetric water contents.

3.6.2 *Moisture retentivity cores*

Retentivity cores were collected on 24 June 1999 from two depths: 0-5 and 5-10 cm. The cores were collected from the middle of both sampling intervals. Samples were approximately 5 cm in diameter and 1-2 cm in depth. Cores were collected from all control paddocks and one paddock from each irrigation study period. Three cores were taken from each depth. The moisture content at matric potential of –50 cm were determined using the "bubble tower method" (Loveday, 1974).

The cores were trimmed at each end until level with the sampling cylinder and placed on a suction plate. Wetting of the soils was undertaken by standing the cores in water, less deep than the lengths of the cores, overnight. When the soil reached saturation, drainage to a soil moisture suction of –50 cm was carried out using bubble tower control. The suction was set to –50 cm by lowering the free water surface in which the bubble tower terminates. At regular intervals, becoming longer as equilibrium was approached, the free water surface was lowered to restore the desired suction. Equilibrium was determined when there was no change in water level over a 12 hour period. The samples were removed from the suction plate, weighed, dried in an oven at 105^{O}C for 24 hours and reweighed. Volumetric water contents were than calculated.

3.6.3 Bulk density

Soil dry bulk density is the ratio of the mass of dry soil to the total volume of soil. Soil bulk density was measured using the "core technique" described by Gradwell (1972) and McLaren and Cameron (1996). Samples were collected on the 6-8 July 1999 from three depths (0-5, 5-10, and 10-15 cm), using cylindrical aluminium cores measuring 5 cm long with an internal diameter of 4.8 cm (mean volume of 90 cm^3). The control paddock from each farm was sampled along with one paddock from every irrigation study period. Sampling sites in each paddock were randomly selected and four samples were collected from each depth in each paddock. The aluminium cores were placed on the soil surface (sharp edge facing the soil) and pushed right into the soil. Using a hand spade, the cores were removed with the intact soil. The soil from the outside of the corer was removed and the soil at both ends of the corer was cut so that the soil surface was flush with the ends of the corer. Cores were placed in sealable plastic bags in order to collect any soil dislodged from the corer during the transport. The cores were oven-dried at 105^0C for 24 hours.

Bulk density measurements were used to allow the soil chemical results to be expressed on a volume basis. As soil chemical results were undertaken to a depth of 60 cm, bulk density measurements were taken at depths down the soil profile. While these samples were unlikely to show any effect of irrigation and/or grazing on bulk density values (due to the depth from the soil surface), they were necessary to compare soil chemical results at different depths. Samples were collected on 23 March 2000 from three depths (20-25, 35-40, and 50-55 cm) for bulk density analysis. The same paddocks were sampled as those on the 6-8 July 1999. Four samples were collected from each depth in each paddock.

3.6.4 Soil infiltration rates

Soil infiltration rates were measured using the "single ring" technique described by Gradwell (1972) and McLaren and Cameron (1996). Soil infiltration rate measurements were taken on 21-23 June 1999. The control paddock from each farm was sampled along with one paddock for every irrigation study period. Five to eight measurements were taken from each paddock.

The infiltration rings were laid out in a diagonal line across the paddock. The surface vegetation was clipped and a cylindrical steel ring (diameter of 400 mm) pushed 4-5 cm into the soil surface. Inside the ring water was ponded on the soil surface and the decrease in water level recorded over time. Measurements were repeated periodically until steady state was achieved. All infiltration rates were measured over a period of at least 2 hours 20 minutes. Soil infiltration status of each paddock is based on the classes in Table 3.5.

Table 3.5 Soil infiltration classes (After Bowler, 1980)

Permeability class	Description	Infiltration rate (mm hr^{-1})
1	Very slow	<1
2	Slow	1-4
3	Moderately slow	5-19
4	Moderate	20-64
5	Moderately rapid	65-129
6	Rapid	130-250
7	Very rapid	>250

3.6.5 Soil strength

Soil penetrometer readings were taken on 10 September 1999 from all study paddocks. A Procter penetrometer was used to measure soil strength. The penetrometer works by recording the force required to push the tip of the penetrometer into the soil. Measurements were taken at random locations on the soil surface. Approximately forty readings were taken per paddock. Different size foots were used for the penetrometer depending on the softness of the paddocks. Generally, the 1.875 or 2.5 cm foot was used on irrigated study paddocks and the 1.25 cm foot used on non-irrigated study paddocks. Soil samples for volumetric soil moisture contents at 0-5 cm depth were collected at the same time.

3.6.6 Aggregate stability

Samples for aggregate stability analysis were collected on 8 November 1999 from all study paddocks at 0-10 cm depth. Field samples were collected using a spade to cut a cube of soil with sides that were approximately 15-20 cm in length. Four sods from each depth in each paddock were put in plastic bags and returned to the laboratory. Samples were partially broken up by hand and air-dried.

Sods were sieved in a mechanical sieve for 10 minutes to obtain an aggregate size between 2 and 5 mm. The sample quantity was then reduced to approximately 500 g using a sample splitter. Moisture contents of all samples were taken at this point. Moisture contents ranged from 15 to 60%, with paddocks irrigated for 40 years having the highest moisture contents. To minimise the variation in moisture contents, the soil samples were air-dried in shallow trays at 30^{0}C overnight. After air-drying, moisture contents were reasonably consistent, ranging from 5 to 9%. Samples were hand sieved to collect aggregates in the range between 2 and 4 mm in diameter. Water-stability of soil aggregates was determined using the technique described by Gradwell (1972) and Kemper and Rosenau (1986). Aggregates were placed on a sieve set and immersed in water and raised and lowered to a depth of 4 cm, 30 times per minute, for 30 minutes. Each sieve was oven-dried at 105^{0}C and the amount of soil in each sieve weighed.

The mass of aggregates retained as a percentage of the total weight of soil added was calculated as

$$\text{Water stable aggregate \%} = \sum[100(a_i - b_i)/((c/mf)\text{-}s)]$$

Where a_i = mass of soil on sieve I, b_i = mass of sand on sieve II, c = mass of air dry soil added, mf = moisture factor, s = total mass of sand in the sample.

3.7 Soil chemical measurements

Soil samples for chemical analyses were collected on the 26-28 of April 1999 from all study paddocks. Samples were collected with a soil corer from five depths: 0-7.5, 7.5-15, 15-30, 30-45, and 45-60 cm. Sixteen cores were collected from each paddock for the first two sampling intervals, and eight cores collected from each paddock for the other three sampling intervals. To collect samples below 30 cm, eight randomly selected holes per paddock were dug to 30 cm and the soil corer inserted to a depth of 60 cm. Samples were put in plastic bags and stored at 4°C until analysis. Samples were passed through a 2 mm sieve and air-dried before analysis.

Results are reported on an oven-dry basis unless otherwise stated. In order to convert results to an oven-dry basis, a moisture factor was applied in the calculation of results.

All samples were analysed for soil pH, total carbon, easily oxidisable organic carbon, total nitrogen, nitrate-nitrogen, ammonium-nitrogen, Olsen P, Phosphate-retention, cation exchange capacity and exchangeable calcium, magnesium, sodium and potassium. The 9 and 14 year study paddocks were also analysed for total phosphorous.

3.7.1 Soil pH

Soil pH was determined in a water suspension of a soil:solution ratio of 1:2.5 (Blakemore *et al.*, 1987). Samples were stirred for 10 minutes and allowed to stand overnight. Measurements were made using a glass/reference electrode.

3.7.2 Carbon

Total soil carbon was determined by the LECO FP 2000 automated analyser, which measures carbon by combustion in a resistance furnace (Blakemore *et al.*, 1987). The sample (200-300 mg finely ground <0.25mm soil) is introduced into the furnace, where it is heated to 1050^0C in a stream of pure oxygen, converting carbon to CO_2. After passing through a thermoelectric cooler, which removes water, the resulting combustion gases are collected in a ballast tank. A sample of gas passes through an infrared cell, which determines the carbon content.

The easily oxidisable organic carbon was determined by the colorimetric method of Walkley and Black (1934) with minor changes (Blakemore *et al.*, 1987). A 0.2 g sample of soil was placed in a 200 ml flask, 12 ml of H_2SO_4 added, mixed and left to stand for 10 minutes. Six ml of 3M chromium trioxide solution was added and the mixture was left to stand for a further 10 minutes. The solution was then diluted with water and left to stand overnight. The solution was then made up to volume with water and well stirred. An aliquot of the solution was centrifuged for 15 minutes at 2000 rpm before being read on a spectrophotometer at 600 nm.

3.7.3 Nitrogen

Total soil nitrogen was determined by the LECO FP 2000 automated analyser (Blakemore *et al.*, 1987). The sample (200-300 mg finely ground <0.25mm soil) is introduced into the furnace, where it is heated to 1050^0C in a stream of pure oxygen, converting nitrogen to N_2 and NOx. After passing through a thermoelectric cooler, which removes water, the resulting combustion gases are collected in a ballast tank. A sample of gas passes through a thermal conductivity cell, which determines the nitrogen content. A heated copper catalyst reduces NOx to N_2, Lecosorb (NaOH on an inert base) removes CO2 and anhydrone (Magnesium perchlorate) removes H_2O, Nitrogen content is then determined by thermal conductivity.

Both nitrate and ammonium were also analysed. A 3 g air-dry soil sample was placed in a Oak Ridge tube and 30 ml of 2M KCl added to the soil. The mixture was shaken for 1 hour. The samples were centrifuged at 8000 rpm for 3 minutes and the solution filtered through Whatman no. 41 filter paper. Soil nitrate and ammonium were determined by AutoAnalyser method (Blakemore *et al.*, 1987) adapted from that of Kamphake *et al.* (1967).

3.7.4 Cation exchange capacity

Cation exchange capacity was determined by leaching with 1 M ammonium acetate (pH 7). 3 g of soil was mixed with 2 g of silica sand and put in micro-columns. The columns were leached with 50 ml of 1 M NH_4oAc. The exchangeable cations (Ca, Mg, K and Na) in the leachate were measured using an atomic adsorption spectrophotometer. The concentration of exchangeable H^+ ions was measured from the changes in pH after leaching in the NH_4oAc.

3.7.5 Phosphorus

Olsen P was determined using the Olsen-soluble phosphorus method (Blakemore *et al.*, 1987). A 1 g sample of soil was placed into a centrifuge tube and 20 ml of 0.5M $NaHCO_3$ (pH 8.5) was added to the soil. The mixture was shaken on an end-over-end shaker for 30 minutes and then centrifuged at 9000 rpm for 1 minute. The solution was decanted off and filtered under suction through a Watman No. 6 filter paper. A 4 ml aliquot of the filtrate was then placed into a 50 ml volumetric flask. Thirty two ml of water was added followed by 10 ml of Murphy and Riley solution (Murphy and Riley, 1962). The solution was then made to volume with distilled water and left for 30 minutes before being read on a UV/visible spectrophotometer at 712 nm.

Total P was determined by Kjedahl digestion of the soils followed by the analysis of P using the molybdenum blue method.

3.7.6 Phosphate retention

Phosphate retention was determined by the method originally devised by Saunders (1965) and revised by Blakemore *et al.* (1987). A 5 g sample of soil was weighed into a centrifuge tube, 25 ml of 1000 mg L^{-1} P (pH 4.6) solution was added and then shaken in an end over end shaker for 16 hours. The tubes were then centrifuged at 8 000 for 5 minutes. A 2 ml aliquot of solution was pipetted into a 50 ml volumetric flask and 12.5 ml of nitric-vanomolybdate reagent was added and the solution made to volume with distilled water. The solution was then left for 30 minutes and read on a UV/visible spectrophotometer at 420 nm.

3.8 Soil solution phosphorus

Field moist soil samples for extraction and analyses of soil solution were collected from all study paddocks on 10-15 September 1999. Soil samples were collected using a soil corer from five depths: 0-7.5, 7.5-15, 15-30, 30-45, and 45-60 cm. Sixteen cores were collected from each paddock for the first two sampling intervals and eight cores were collected from each paddock for the other three sampling intervals. To collect samples below 30 cm, four randomly selected holes were dug to 30 cm and the soil corer inserted to 60 cm. Samples were put in plastic bags and stored at 4^{O}C until extraction occurred within 4 days. Soil solution were removed from the field moist soil by centrifugation at 10 000 rpm in a refrigerated centrifuge at 5^{O}C (Elkhatib *et al.*, 1987). Soil solution was analysed for P.

3.9 Suction cups

Suction cups were installed on 30 August 1999 in paddocks 2 (non-irrigated) and 16 (irrigated for 40 years) on farm No.2 (Figure 3.5). The suction cups were installed at two depths: 15 and 60 cm. Five suction cups were installed at each depth in each paddock, giving a total of 20 suction cups. The suction cups were installed by augering a hole with a screw auger to the required depth and then carefully inserting the suction cup. A soil and water slurry was then poured into the hole to ensure good contact between the soil and the porous cup. The suction cups were fenced off from stock to stop breakage and direct leaching of nutrients from urine and dung patches.

Figure 3.5 *Suction cups in paddock no.2 on Farm no.2*

Suction cup samples were collected on four different occasions: 17 September 1999, 12 November 1999, 5 January 2000, and 29 January 2000. Samples were collected after paddock 16 was irrigated with effluent. A garden watering can was used to irrigate the suction cups in paddock 2, with approximately 35 mm of water. A suction of 70-80 kPa was set on the suction cups for 24 hours and the soil solution sample was then collected by syringe. The samples were collected in 100 ml plastic containers and either stored at $4^{O}C$ or frozen until analysis was undertaken. The samples were analysed for P and N.

3.10 Phosphate adsorption isotherms

Soil samples collected for the soil chemical measurements (section 2.5) were also used for the phosphate adsorption isotherms. The effect of the duration of effluent application on P adsorption was examined by measuring P adsorption for 0-7.5 cm depth soils from all paddocks. The effect of soil depth on P adsorption was examined by measuring P adsorption at all depths for the 40 year application paddocks. Soil (1g) was shaken with a solution of 0.1 M NaCl containing varying amounts of phosphate added as KH_2PO_4. One drop of AR chloroform was added to the samples and these were shaken on an end-over-end shaker for 40 h at 20^0C at a solution:soil rate of 40:1. At the end of the shaking period, the solution was separated by centrifugation and filtration. Phosphate concentrations were measured by the method of Murphy and Riley (1962). The amount of P adsorption was calculated from the difference between the concentrations of P in the original solution and that remaining after adsorption.

3.11 Soil biological measurements

Soil biological measurements included soil respiration rates and earthworm populations.

3.11.1 Earthworm populations

Earthworm populations were determined using the "technical spade" method described by Gilyarov (1975). Soil samples were collected at the beginning of September 1999, from all control paddocks and two of the 9, 30 and 40 study paddocks. Samples were collected from six randomly selected locations within each of the paddocks. Fraser *et al.* (1999) found the variability in earthworm population estimates did not decrease with an increase in the number of replicates and hence a small number of replicate samples is adequate to provide an estimate of the true earthworm population size. Using a spade, a block of soil with approximate dimensions of 16 cm x 16 cm x 9 cm deep was collected and placed in a plastic bag. Each sample was returned to the laboratory, where it was carefully hand sorted to collect all resident earthworms. The most reliable method for measuring earthworm populations is careful hand sorting of earthworms from a block of soil of known volume (Fraser *et al.*, 1999). All earthworms recovered were counted and weighed. Numbers of earthworms found in each sample were converted to an area basis (number of worms m^{-2}) and mean individual earthworm mass calculated.

3.11.2 Soil respiration rates

Soil samples for respiration rates were collected on 23 June 1999 at 0-10 cm depth, from all study paddocks. Samples were collected with a soil corer, bulked together, placed in plastic bags and stored at $4^{O}C$ until analysis. Basal soil respiration rates and relative use of glucose substrate by microbial communities in each soil sample (over a short period of time) were determined using the "Anderson and Domsch " Glucose amendment technique (Anderson and Domsch, 1977). This technique quantifies soil microbial biomass through its response to substrate addition. Soil (20 g) was mixed with 1.0 g of glucose and sealed into a gas tight flask (100 ml) and incubated at room temperature for 24 hours with a CO_2 trap (KOH). The accumulation of respired CO_2 was determined by back titration with HCl. Absorbance of atmospheric CO_2 was determined by titration of blanks of fresh KOH with HCl. Substrate use was calculated as the difference between CO_2-C in the substrate and the blank treatments.

3.12 Grazing trial

A grazing trial was designed to assess the impact of grazing on some soil physical characteristics of recently irrigated pasture. A paddock irrigated for 40 years (paddock 7, farm No. 2) was selected for the trial because it had consistently higher moisture contents than other irrigated paddocks. For this reason, paddocks irrigated for 40 years are more likely to be susceptible to treading damage and changes in soil physical characteristics. The stocking rate used for the grazing trial was 3.2 cows ha^{-1}.

Paddock seven was fenced in half providing two treatments:

Treatment 1 (T1) - one irrigation run over half the paddock (34 millimetres of effluent was applied) with 12 hour grazing the following day.

Treatment 2 (T2) - two irrigation runs over the other half of the paddock (74 millimetres of effluent was applied) with 12 hour grazing the following day.

Application doses were measured by collecting effluent in several randomly located containers along the irrigation run.

A number of soil physical measurements were made before (used as the control) and after irrigation/grazing including: soil moisture content, soil bulk density, soil infiltration rate, and surface roughness.

3.12.1 Bulk density

Bulk density samples were collected 2 days before grazing (T1 - 29/10/1999 and T2 - 1/11/1999) and 1 day after grazing (T1 - 1/11/1999 and T2 - 5/11/1999). Samples were collected from two depths: 0-5 and 5-10 cm. Twelve random samples were collected from each depth. Cores were placed in sealable plastic bags before analysis. Samples were analysed using the method described in Section 3.6.3.

3.12.2 Soil infiltration rate

Soil infiltration rates were measured 1 day before irrigation (T1 - 29/10/1999 and T2 - 1/11/1999) and 3 days after irrigation (T1 - 2/11/1999 and T2 - 5/11/1999). Five to eight measurements were taken from each paddock. The rings were layed out in a diagonal line across the paddock. All infiltration rates were measured over a period of at least 2 hours. Measurements were taken using the method described in section 3.6.4.

3.12.3 Surface roughness

To assess the effect on soil surface roughness of stock grazing on recently irrigated pasture, surface roughness was measured before and after grazing. Surface roughness was measured by using the roller chain method. Measurements were made 1 day before irrigation (T1 - 29/10/1999 and T2 - 1/11/1999) and 1 day after grazing (T1 - 1/11/1999 and T2- 5/11/1999). Twelve random measurements were recorded for each treatment.

The procedure involves laying a roller chain (1.04 m in length) on the ground in a straight line and moulding it to the surface contour. The rougher the surface the greater the proportional loss in length of the chain compared to if it were on a flat surface. Surface roughness can be assigned to a roughness class depending on the amount of apparent loss in chain length as shown in Table 3.6.

Table 3.6 Surface roughness classes

Loss in length of chain (%)	Roughness class	Roughness name
0 – 5	1	Slightly rough
5 – 10	2	Moderately rough
10 – 15	3	Distinctly rough
15 – 20	4	Very rough
>20	5	Extremely rough

3.13 Bare patches

Bare patches have developed in some irrigated paddocks, particularly on farm No. 2 (Figure 3.6). The bare patches range in size from approximately 5 m^2 to 20 m^2 and are characterised by soft, dark topsoil approximately 0-15 cm thick overlying a considerably harder, lighter coloured layer. These areas have only a small amount of vegetation growing on them. The vegetation is easily removed by pulling and would be very susceptible to 'grass pulling' by stock. Soil samples from 4 bare patches were collected for chemical and physical analysis to characterise these areas. Two bare patches in paddock 7 and two in paddock 16 on farm No.2 were selected for sampling.

Figure 3.6 *Bare patches developed in paddocks 16 (left) and 7 (right) on Farm No. 2*

3.13.1 Bulk density

Bulk density samples were randomly collected on 9-11 November 1999 from all 4 bare patches at three depths: the middle of the 0-15, 15-20 and 20-25 cm horizons. Five samples were collected from each depth. Samples were analysed using the method described in Section 3.6.3.

3.13.2 Chemical analysis

Soil samples for chemical analyses were collected on the 9-11 November 1999 from all 4 bare patches at five depths: 0-5, 5-10, 10-15, 15-20, 20-25 and 25-30 cm. A soil corer was used to collect the soil samples. Samples were put through a 2 mm space sieve and air-dried before analysis. Samples were analysed for total carbon, Olsen P, P retention, cation exchange capacity, exchangeable calcium, magnesium, sodium and potassium and sulphate sulphur. All analyses were undertaken using the methods described in Section 3.7, except for sulphate sulphur, which is described below.

3.13.3 Phosphate-extractable sulphate sulphur

Phosphate-extractable sulphate was determined by the method described by Blakemore *et al.* (1987). A 5 g (air-dried, <2 mm) sample was placed into a centrifuge tube and 25 ml of extracting solution 0.04 M $Ca(H_2PO_4)_2$ (Searle, 1979) was added. The solution was shaken end over end for one hour and centrifuged at 2000 r.p.m. for 15 minutes. Two ml aliquots of sample extracts were transferred to 50 ml flasks and 6 ml of reduction mixture added before distillation was carried out. After distillation, 10 ml of colour reagent was added and the solution shaken, followed by the addition of 2 ml of ammonium ferric sulphate reagent. The flasks were made up to volume with water and left to stand for 30 minutes before being read at 670 nm on a spectrophotometer (Johnson and Nishita, 1952).

3.14 Pasture Analysis

Pasture samples were collected on 12 November 1999 from all of the study paddocks. Approximately 30 random cuts were taken at grazing height (with shears) per paddock and placed in a bucket. A smaller representative amount was subsampled and immediately placed in a chilly bin with ice. Samples were microwaved on high for 3 minutes before being placed on ice and transported to the laboratory for analyses. Samples were oven dried at 60^0 C for 24 hours and put in airtight containers. AgResearch received the samples on 16 November 1999 for analysis.

Samples were analysed for protein, lipid, ash, fibre, dietary anion-cation difference, soluble sugars and starch, organic matter digestibility and estimated metabolisable energy. The method used for analysis was near-infra red reflectance spectrometry prediction using calibrations based on wet chemistry methods.

3.15 Historical sacrifice area

Historically, paddock 14 on farm No.2 has been subjected to excessive loadings of effluent. The stationary sprinkler mounted on a trolley was left unattended in this paddock for extended periods of time. The concept of 'out of sight out of mind' by the sharemilker may have been the reason for the neglect of this sprinkler. In reality, the paddock was used as a sacrifice area for the disposal of effluent, rather than the treatment of effluent. The paddock was very wet, and often suffered from ponding and pugging problems as well as weed infestation and bare patches.

It is very difficult to estimate how much extra irrigation this paddock has received. It is unlikely that, in the future, irrigated paddocks will suffer the problems experienced in paddock 14. Paddock 14 had several years of bad management that may equate to decades of management under the new improved regime, in terms of nutrient loadings. Analyses of soil samples from paddock 14 may give an indication of the nutrient levels that can be expected at some point in the future in other paddocks receiving effluent irrigation.

Soil samples for chemical analysis from this paddock were collected on 8 November 1999. Samples were collected with a soil corer from five depths: 0-7.5, 7.5-15, 15-30, 30-45, and 45-60 cm. Sixteen cores were collected from the paddock for the first two sampling depths and eight cores collected from the paddock for the other three depths. To collect samples below 30 cm, eight random holes, were dug to 30 cm and the soil corer inserted to 60 cm. Samples were put through a 2 mm space sieve and air-dried before analysis. Sub-samples were taken from the soil samples to determine volumetric moisture contents.

Soil samples were analysed for total carbon, Olsen P, P retention, cation exchange capacity, exchangeable calcium, magnesium, sodium and potassium and sulphate sulphur. All analyses were undertaken using the methods described in sections 3.7 - and 3.13.2.

3.16 Statistical Analysis

The SAS statistical programme was used to perform analysis of variance and selected correlations. The analysis of variance was used to compare the difference between the five treatment means. No analysis was undertaken comparing means of different soil depths. Where unusual differences between soil depths occurred, these were discussed.

CHAPTER 4

Results and Discussion

4.1 Introduction

The results section covers all aspects of the characterisation of the soil resource, i.e. physical, chemical and biological measurements. Attempts are made to identify significant results and trends and ascertain the factors and interactions responsible for the results. Earlier water quality data collected by Taranaki Regional Council (TRC), some soil chemical analysis undertaken by New Zealand Dairy Research Institute (NZDRI) and pasture analysis carried out by AgResearch (AgR) for the site are also presented and discussed.

4.2 Effluent characteristics

Effluent samples were collected from the pumphouse located adjacent to the effluent balance tank. Three separate samples were collected at regular time intervals over a six hour period, which is approximately the time required to complete one irrigation run by a travelling irrigator. The results of the analysis are shown in Table 4.1.

The effluent analysis results show there is extreme variability over short time intervals for all commonly measured effluent properties. Previous analysis of LNZ effluent have shown similar variability (NZDRI, 1993). LNZ effluent is characterised by high suspended solids, BOD_5, COD, total phosphorus, total potassium and total sodium and low pH. In comparison to other commonly irrigated wastes in New Zealand (Table 2.1), lactose effluent is somewhat distinct. It has higher BOD_5 and COD levels than both dairy factory and meat processing effluent, a lower pH and similar levels of total nitrogen and phosphorus.

Table 4.1 Lactose effluent composition

Test	Units	Start of Run 9.00 a.m.	Middle of Run 12.00 p.m.	End of Run 3.00 p.m.
pH	pH units	4.4	4.3	4.3
Electrical conductivity*	m Sm^{-1}		2.76 @ $38^{O}C$	
Total Suspended Solids	$g.m^{-3}$	364	578	552
Total Carbon**	%		38.7	
Total Solids***	%		1.77	
Total Sodium	$g.m^{-3}$	78.7	181	172
Total Potassium	$g.m^{-3}$	220	367	354
Total Ammoniacal-N	$g.m^{-3}$	6.7	21	17.9
Total Kjeldahl Nitrogen (TKN)	$g.m^{-3}$	57.8	123	104
Nitrate-N + Nitrite-N (TON)	$g.m^{-3}$	9.49	60.4	54.9
Nitrate-N	$g.m^{-3}$	9.49	60.4	54.9
Nitrite-N	$g.m^{-3}$	< 0.002	0.002	0.004
Total Phosphorus	$g.m^{-3}$	72.1	100	91
Sulphate	$g.m^{-3}$	20	33	31
Carbonaceous Biochemical Oxygen Demand (CBOD~5^)	g.O~2^.m^{-3}	19 100	10 200	7 200
Total COD	g.O~2^.m^{-3}	28 700	17 300	12 100

* Electrical conductivity was analysed separately by LNZ at their laboratory using a 'grab' effluent sample.
** Total carbon was determined at Massey University by drying an effluent sample in an oven at $105^{O}C$, recovering the solids remaining and analysing using a Leco furnace.
*** Total solid content of the effluent was determined on a volume basis at Massey University by drying an effluent sample in an oven at $105^{O}C$.

The composition of the effluent has altered considerably over time. Results of analysis performed on the lactose effluent either by LNZ or the Taranaki Regional Council are presented in Table 4.2.

Table 4.2. Changes in effluent composition over time (units are the same as in Table 4.1)

Test	1992	1994	1995	1996	1997	1997
COD	3 400	11 000	3900	3930	7 200	15 400
PH	6.9	6.6	4.1	4.2	4.5	4.1
Suspended Solids	400	-	291	97	300	547
Total nitrogen	155	-	54	55	60	127
Total phosphorus	80	-	39.3	101	80	117

The data in Table 4.2 indicates LNZ effluent has changed from near neutral pH to a relatively acidic pH. COD is still highly variable but may have increased in overall concentration, suspended solids may have increased slightly. This may be related to the introduction of carbon in the filtering process. The changes in the concentrations of total nitrogen and phosphorus are not consistent.

Using the concentrations in Table 4.1, a land treatment area of 120 ha and an approximate effluent volume of 600 000 m^3 irrigated annually, the approximate annual nutrient loadings were calculated and are shown in Table 4.3.

Table 4.3. Total annual nutrient loadings and nutrient loading ha^{-1}

Test	Total annual nutrient loading-average (tonnes)	Total annual nutrient loading-range (tonnes)	Nutrient loading rate-average (kg ha^{-1})	Nutrient loading rate-range (kg ha^{-1})
Total Suspended Solids	300	218.40 – 346.80	2 490	1820 – 2890
Total Sodium	86.34	47.22 – 108.60	720	395 – 905
Total Potassium	188.22	132.00 – 220.20	1 570	1100 – 1835
Total Nitrogen	56.94	34.68 – 73.80	475	290 – 615
Total Phosphorus	52.26	43.26 – 60.00	435	360 – 500
Sulphate	16.80	12.00 – 19.80	140	100 – 165

Table 4.3 shows that extremely high rates of sodium, potassium, phosphorus and suspended solids are being applied through irrigation. This is a concern for both soil and water quality. Sodium ions have the ability to cause degradation of soil structure and strength through the deflocculation of clay particles. This can lead to a reduction in soil infiltration rates and an increase in the occurrence of pugging by grazing animals. High concentrations of sodium and potassium is likely to induce the leaching of Ca and Mg in soil, resulting in Ca and Mg deficiencies in pasture and the grazing animal. Sodium ions are weakly held on soil exchange sites and are very susceptible to leaching. High concentrations of sodium have been reported in the groundwater below the land treatment area. Concentrations have been measured at 179 g m^{-3} below farm No. 1 and 129 g m^{-3} below farm No. 2 (TRC, 1999).

The Petrochem Ltd Ammonia Urea plant, situated at Kapuni, to the east of LNZ, spray irrigates effluent to both a 'cut and carry' and 'grazed' land treatment areas (Stevens, 2000). The resource conditions specify a 1000 kg N ha^{-1} yr^{-1} loading limit for the cut and carry area and a 300 kg N ha^{-1} yr^{-1} limit for the grazed area. However, Petrochem aim to manage the loading rates such that they do not exceed 600 and 200 kg N ha^{-1} yr^{-1}, respectively (Stevens, 2000). Degens *et al.* (2000) reported on average over a 22 year period 570 kg P ha^{-1} yr^{-1}, 960 kg N ha^{-1} yr^{-1} and 14.7 tonnes organic C ha^{-1} yr^{-1} were applied to an allophanic soil used for dairy farming in the Waikato, through irrigation of a dairy factory effluent. Keeley and Quin (1979) reported very large application loading rates of nutrients, including 920 kg N ha^{-1} yr^{-1}, 2120 kg Na ha^{-1} yr^{-1}, and 830 kg K ha^{-1} yr^{-1} with the application of meatworks-fellmongery effluent at Fairton in Canterbury. Dairy effluent land application systems are normally designed to meet loading rates of 150-200 kg N ha^{-1} yr^{-1}, which equates to the application of approximately 300-400 kg K ha^{-1} yr^{-1}.

LNZ has moderate application rates of nitrogen and moderate to high application rates for suspended solids, phosphorus, sodium and potassium. The loading rates are higher than the recommended application rates or those required by pasture.

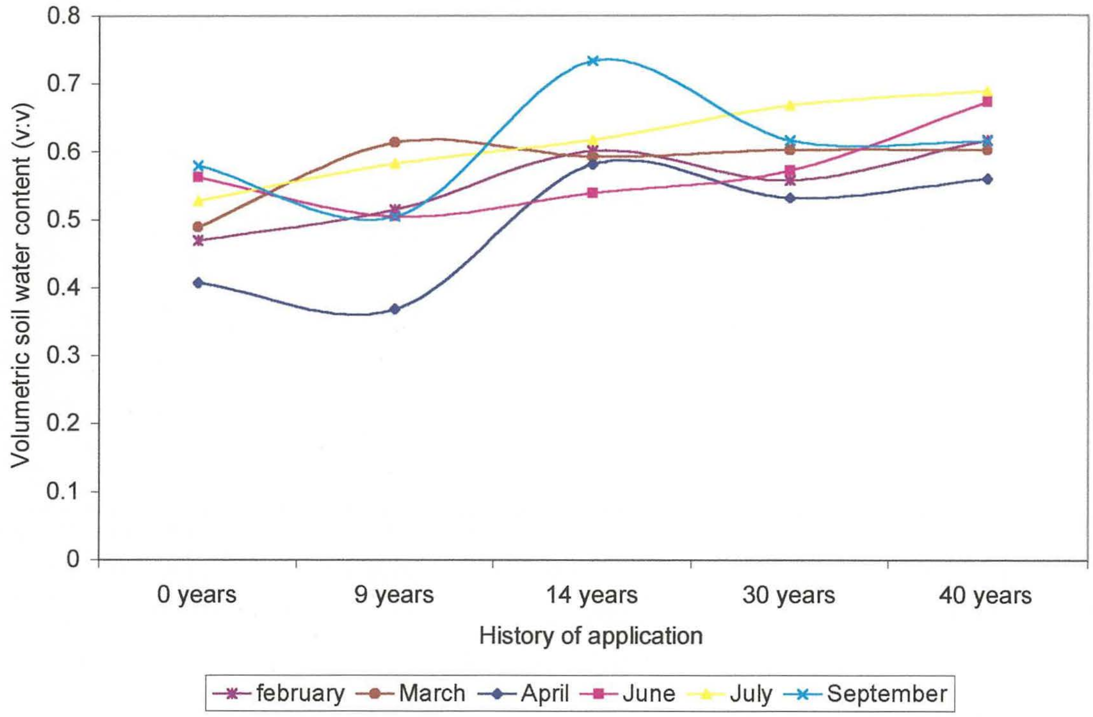
4.3 Effect of effluent on soil physical properties

The irrigation of effluent has altered some of the physical characteristics of the irrigated soil. The most appreciable changes have occurred at the 0-7.5 cm soil depth, with the effects diminishing with increasing soil depth. The effects of effluent irrigation on soil physical properties increased with increasing period of irrigation.

4.3.1 Soil moisture content

Soil moisture contents in the 14, 30, and 40 study irrigated paddocks were consistently greater than those in the non-irrigated study paddocks throughout the year (Figure 4.1). For some of the sampling periods the 9 year study paddocks had soil moisture contents similar to the non-irrigated study paddocks. There is a general trend of increasing soil moisture content with increasing years of effluent irrigation. The 14 year study paddock when sampled in April, September and February had relatively high moisture contents. This can be attributed to what resembles an iron pan in the subsoil, restricting permeability. It is also possible that these paddocks may have been irrigated in the preceding day(s), before the samples were collected.

Figure 4.1 *Volumetric soil water content for 0-7.5 cm depth with varying years of effluent irrigation*



Degens *et al.* (2000) noted soil moisture content to a depth of 50 cm, was greater under pasture irrigated with dairy factory effluent for 22 years, than non-irrigated pasture. For example, at 0-10 cm depth, non-irrigated soil had a volumetric moisture content of 54% compared with 75% for irrigated soil.

On two occasions (26-28 April 1999 and 10-15 September 1999) soil samples were collected from 7.5-15, 15-30, 30-45 and 45-60 cm depth and volumetric moisture contents determined. The results show irrigated study paddocks have considerably greater soil moisture contents than non-irrigated study paddocks at the 0-15 cm depth. The greatest increases in soil moisture contents were measured at the 7.5-15 cm depth. To a depth of 30 cm, there is a general trend of increasing soil moisture content with increasing years of effluent irrigation. Below 30 cm, effluent irrigation has had little effect on the soil moisture content.

4.3.2 Soil moisture retentivity cores

Moisture retentivity cores were collected from all study paddocks at two depths (0-5 and 5-10 cm) and the volumetric moisture content at matric potential of –50 cm determined. Although the irrigated paddocks had a higher volumetric water content than the non-irrigated paddocks, the increase was significant only for the 40 year study paddocks (Table 4.4). Soil moisture contents for 9, 30 and 40 year study paddocks were greater ($P \leq 0.05$) than the non-irrigated paddocks at 5-10 cm depth. Soil water holding capacity can be increased by the input of organic matter in the applied waste and by the improved soil physical conditions (Mbagwu and Piccolo 1990).

Table 4.4 Volumetric soil moisture contents at –50 cm for soils receiving varying years of effluent application (mean ± standard deviation)

Soil Depth	0 Years	9 Years	14 Years	30 Years	40 Years
0-5 cm	$0.65_a \pm 0.04$	$0.68_a \pm 0.08$	$0.74_{ab} \pm 0.05$	$0.76_{ab} \pm 0.09$	$0.88_b \pm 0.08$
5-10 cm	$0.56_a \pm 0.05$	$0.71_{bc} \pm 0.03$	$0.63_{ab} \pm 0.05$	$0.79_c \pm 0.04$	$0.81_c \pm 0.06$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

Regression analysis determined a strong relationship between volumetric moisture content at –50 cm and years of effluent irrigation ($r^2 = 0.91$ at 0-5 cm and 0.85 at 5-10 cm depth).

4.3.3 Soil bulk density

The bulk density of the topsoil in the irrigated paddocks is very low (Table 4.5). Bulk density values have decreased in the irrigated paddocks at the 0-15 cm soil depth compared with the non-irrigated paddocks. There is a strong trend of decreasing soil bulk density with increasing years of effluent irrigation at 0-15 cm soil depth. As expected, this is most apparent at 0-5 cm depth, where the greatest decreases in bulk density were measured in all study paddocks. Bulk density values are significantly lower ($P \leq 0.05$) on all irrigated study paddocks compared to non-irrigated study paddocks at 0-5 cm depth. At the 10-15 cm depth, only the 40 year study paddock has bulk density values significantly lower than the non-irrigated study paddocks. Below 20 cm, bulk density values are very similar across all study years and sampling depths. The 14 year study paddock has bulk density values slightly greater than all other study year paddocks below a depth of 20 cm. This can be attributed to the iron pan that exists in the subsoil of these paddocks. This may explain some of the anomalies in chemical data for this period of effluent irrigation (Section 4.4).

Table 4.5 Bulk density values for soils receiving varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	0 Years ($g\ cm^{-3}$)	9 Years ($g\ cm^{-3}$)	14 Years ($g\ cm^{-3}$)	30 Years ($g\ cm^{-3}$)	40 Years ($g\ cm^{-3}$)
0-5 cm	$0.89_a \pm 0.04$	$0.67_b \pm 0.04$	$0.64_b \pm 0.03$	$0.61_b \pm 0.04$	$0.52_c \pm 0.02$
5-10 cm	$0.95_a \pm 0.07$	$0.84_b \pm 0.02$	$0.88_a \pm 0.04$	$0.76_b \pm 0.02$	$0.68_b \pm 0.04$
10-15 cm	$0.95_a \pm 0.05$	$0.94_a \pm 0.04$	$0.91_a \pm 0.08$	$0.87_a \pm 0.05$	$0.81_b \pm 0.09$
20-25 cm	$0.91_{ab} \pm 0.07$	$0.88_{ab} \pm 0.04$	$0.99_a \pm 0.04$	$0.84_b \pm 0.07$	$0.90_{ab} \pm 0.05$
35-40 cm	$0.93_a \pm 0.07$	$0.94_a \pm 0.01$	$0.97_a \pm 0.03$	$0.87_a \pm 0.04$	$0.94_a \pm 0.04$
50-55 cm	$0.94_a \pm 0.07$	$0.96_a \pm 0.04$	$0.98_a \pm 0.02$	$0.91_a \pm 0.05$	$0.96_a \pm 0.03$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

Regression analysis determined a strong negative relationship between bulk density and the number of years paddocks had been irrigated ($r^2 = 0.83$ at 0-5, 0.88 at 5-10 and 0.84 at 10-15 cm). Khaleel *et al.* (1981) attributed a decrease in bulk density to the dilution effect resulting from the mixing of the added organic matter. An increase in aggregation and therefore an increase in porosity would also be expected to have contributed to the decrease in soil bulk density.

The New Zealand Dairy Research Institute has undertaken some bulk density measurements on farms No. 1 and No. 2. The results of the sampling are shown in Table 4.6.

Table 4.6 Bulk density values measured by NZDRI (NZDRI, 1993)

Years of Irrigation (At measurement)	Soil Depth (cm)	Bulk Density (g cm^{-3})
0	0-7.5 cm	0.78
7	0-7.5 cm	0.79
7	25-28 cm	0.84
7	28-60 cm	1.20
7	60-120 cm	1.40
23	0-7.5 cm	0.50
33	0-7.5 cm	0.62

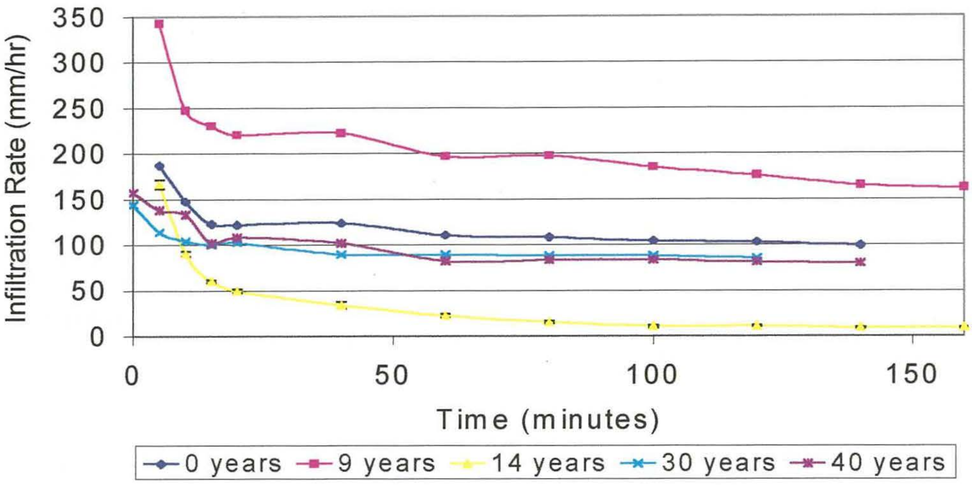
The results in Table 4.6 are reasonably consistent with those reported in Table 4.5, especially at shallower depths. The paddocks irrigated for 23 and 33 years have lower bulk density values than the non-irrigated paddocks. Paddock 9 on farm 1 has a very low bulk density, which NZDRI partially attribute to grass pulling (NZDRI, 1993). NZDRI measured comparatively high bulk density values at the 28-60 and 60-120 cm depth.

McAuliffe (1978) reported a significant decrease in bulk density from 0.95 g cm^{-3} to 0.74 g cm^{-3} on the surface 10 cm at the Te Rehunga dairy factory land treatment site irrigated for 15 years. The decrease in bulk density is similar to that reported for the 14 year study paddock in Table 4.5.

4.3.4 Soil infiltration rates

The irrigation of effluent has had no consistent effect on the infiltration rate of the irrigation study paddocks (Figure 4.2). The non-irrigated, 30 and 40 year paddocks have similar infiltration rates but the 14 year study paddock has a very low infiltration rate and the 9 year paddock has a relatively high infiltration rate.

Figure 4.2 *Average surface infiltration with varying years of effluent irrigation*



Addition of organic matter to soils is generally expected to increase soil permeability, as it contributes to increased porosity and structural stability (Khaleel *et al.*, 1981; Reed and Crites, 1984). However, land application of wastes is reported to have a variable effect on the infiltration rate and hydraulic conductivity of soil (Cameron *et al.*, 1997).

The 14 year study paddock had an infiltration rate of approximately 9 mm hr^{-1}, which was significantly lower ($P \leq 0.05$) than all other study paddocks. This is not unexpected as paddocks on the western side of Dunns Creek (No. 1 Farm) have what resembles an iron pan, which restricts permeability in the subsoil.

The 9 year study paddock had an infiltration rate of approximately 160 mm hr^{-1}, which was significantly greater ($P \leq 0.05$) than all other study paddocks. It is difficult to determine why the 9 year study paddock has such a high infiltration rate. Increases in soil permeability have been reported following additions of effluent (Mathan, 1994). Magesan *et al.* (1996) reported that the application of secondary-treated sewage effluent increased the saturated hydraulic conductivity from 39 to 57 mm hr^{-1}. The 0, 30 and 40 year study paddocks had similar infiltration rates of 95, 80 and 85 mm hr^{-1}, respectively.

Apart from the 14 year paddock, which is classified as having a 'moderately slow' infiltration rate, all other soil infiltration rates are classified by Bowler (1980) as 'moderately rapid'. However, all soil infiltration measurements were made using clean water. The infiltration rates may be lower when factory effluent is used. Effluent may move into and through the soil more slowly than water. Recommendations for the effluent application rate based on the infiltration rate obtained using water vary widely. Vanderholm and Beer (1970) suggested that effluent irrigation rates should be 50-70% of the clean water infiltration rate. Permeability as low as 10 percent of the water value has been reported (USEPA, 1976).

NZDRI staff undertook soil infiltration rate measurements on farms No. 1 and No. 2 in 1987. The average infiltration rates are shown in Table 4.7.

Table 4.7 Soil infiltration rates on farms No. 1 and No. 2 in 1987 (NZDRI, 1993)

Years of irrigation (at 1987)	Years of Irrigation (at 2000)	Number of measurements	Infiltration rate (mm hr^{-1})	Standard deviation
0 years	0 years	4	83.8	57.5
1 year	14 years	2	61.0	19.8
17 years	30 years	10	24.7	12.4
27 years	40 years	6	38.3	31.0

The results in Table 4.7 are generally lower than those presented in Figure 4.2. Some of the differences may be due to the different paddocks sampled and the spatial variation that exists when measuring soil infiltration rates. The non-irrigated paddocks have approximately similar soil infiltration rates; 95 mm hr^{-1} (1999) compared with 84 mm hr^{-1} (1987). Effluent irrigation may have reduced the soil infiltration rates of the irrigated paddocks. There are a number of factors that may have increased soil infiltration rates:

- Hydraulic loading rates have decreased since 1987 due to the addition of farm No.3 and installation of inground sprinklers, to increase the area of land available for irrigation.
- Re-design of irrigation runs to reduce overlap, decreasing hydraulic loadings.

- Amendment of the management philosophy in 1995 aimed at improving the farms long-term sustainability of effluent treatment activities. This includes practices such as, removal of all stock to winter grazing, pasture renewal programmes and regular subsoiling of paddocks.

While there is a lack of consistency with the NZDRI results, there are no significant differences compared to the results in this study.

4.3.5 Soil Strength

Penetration resistance is significantly lower ($P \leq 0.0001$) on irrigated study paddocks compared to non-irrigated paddocks (Table 4.8); a mean value of 16.8 kg f cm^{-2} for irrigated study paddocks compared to 50.5 kg f cm^{-2}. There is a general trend of decreasing penetration resistance with increasing years of irrigation. Greater soil moisture contents of the irrigated paddocks combined with lower bulk density values than the non-irrigated paddocks, mean the irrigated paddocks are softer decreasing the force required to insert the penetrometer. Regression analysis determined a negative relationship between penetration resistance and volumetric soil moisture content ($r^2 = 0.67$). While it is ideal to have all paddocks at similar moisture contents when sampling, it is inconceivable at the LNZ irrigation farms.

Table 4.8 Penetrometer reading from paddocks receiving varying years of effluent irrigation (mean ± standard deviation)

Years of irrigation	0 Years	9 Years	14 Years	30 Years	40 Years
Penetration resistance ('000 kPa)	$49_a \pm 19$	$27_b \pm 7$	$16_c \pm 5$	$11_c \pm 4$	$13_c \pm 3$
Volumetric Moisture Content	0.61	0.71	0.69	0.78	0.80

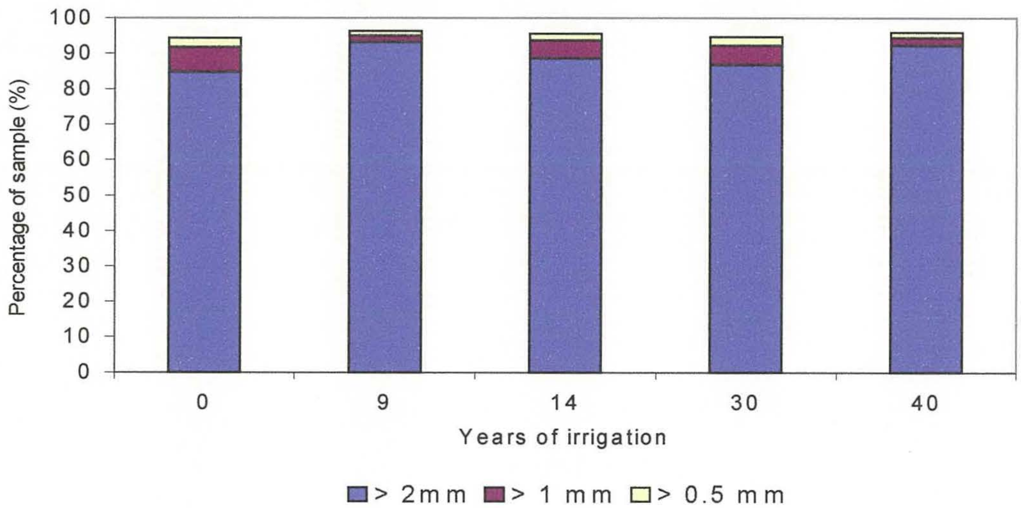
Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

Accumulation of soil organic matter and an increase in soil moisture retention may also lead to a decrease in soil strength, measured by a decrease in penetration resistance (Stevens *et al.*, 1988). A reduction in soil bulk density is also likely to lead to a reduction in soil strength.

4.3.6 Aggregate stability

All irrigated study paddocks had similar or slightly greater aggregate stability than the non-irrigated paddocks (Figure 4.3). When comparing individual fractions, the non-irrigated paddocks had the least stable aggregates greater than 2 mm in size (84.9%). The 9 and 40 year study period paddocks had the most stable aggregates greater than 2 mm, 93.3% and 92.4%, respectively. Consequently, the non-irrigated paddocks had greater percentage of aggregates in the greater than 1 mm and greater than 0.5 mm fractions. There was no trend of increasing aggregate stability with increasing years of irrigation. The only significant difference ($P \leq 0.05$) measured was between the 0 and 9 year study paddocks.

Figure 4.3 *Aggregate stability of 0-10 cm depth for soils with varying years of effluent irrigation*



The effect of the added organic material on aggregate stability depends on the texture of the soil. The Lactose farms have volcanic loam soils with inherently stable aggregates. Therefore, the addition of organic carbon via irrigated effluent has had no noticeable effect on the aggregate stability. Mbagwu and Piccolo (1990) reported the additions of pig slurry, sewage sludge and cattle slurry to a clay soil had no significant effect whilst additions to a sandy loam soil were reported to increase soil aggregate stability by 34, 41 and 26%, respectively.

4.4 Effect of effluent on soil chemical properties

All soil chemical analysis data are expressed on a volume basis due to the large differences in soil bulk density values between the non-irrigated study paddocks and the irrigated study paddocks (Section 4.3.3).

The effluent irrigation has resulted in a marked increase in the level of soil chemical fertility on the land treatment area. Increases of some nutrient levels have not only occurred in the 0-7.5 cm soil depth, but down the soil profile. Generally, there is a trend of increasing nutrient concentrations with increasing years of effluent irrigation.

The New Zealand Dairy Research Institute has been undertaking chemical analysis on soil samples in the 0-7.5 cm depth, from farm No. 1 and No. 2 since 1992 and farm No.3 since 1994 (NZDRI, 1998).

4.3.1 Soil pH

Irrigation of effluent has increased soil pH at all depths sampled (Table 4.9) with a mean value of 6.82 on irrigated paddocks compared with 6.11 on non-irrigated paddocks, at 0-7.5 cm depth. At the 7.5-15 and 30-60 cm soil depth, all irrigation paddocks have significantly greater soil pH ($P \leq 0.05$) than the non-irrigated paddocks. Below 30 cm, soil pH generally increases with increasing years of effluent irrigation. Soil pH generally increases with increasing soil depth on all study paddocks. The 9 year study paddocks have relatively high soil pH over the first three sampling depths.

Table 4.9 Soil pH for soils with varying years of effluent irrigation (mean ± standard deviation)

	Years of effluent irrigation				
Soil Depth	0 years	9 years	14 years	30 years	40 years
0-7.5 cm	$6.11_a \pm 0.12$	$6.97_b \pm 0.09$	$6.86_b \pm 0.04$	$6.59_{ab} \pm 0.40$	$6.88_b \pm 0.06$
7.5-15 cm	$5.88_a \pm 0.18$	$7.06_b \pm 0.04$	$6.87_b \pm 0.03$	$6.78_b \pm 0.09$	$7.08_b \pm 0.09$
15-30 cm	$6.15_a \pm 0.37$	$7.02_{ab} \pm 0.11$	$6.97_{ab} \pm 0.05$	$6.86_{ab} \pm 0.15$	$7.12_b \pm 0.05$
30-45 cm	$6.41_a \pm 0.18$	$7.07_b \pm 0.11$	$7.13_b \pm 0.06$	$7.11_b \pm 0.04$	$7.24_b \pm 0.09$
45-60 cm	$6.62_a \pm 0.11$	$7.09_b \pm 0.10$	$7.17_b \pm 0.08$	$7.27_b \pm 0.08$	$7.28_b \pm 0.03$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI (NZDRI, 1998) indicate soil pH levels in irrigated paddocks are slightly higher than non-irrigated paddocks. Their measurements show soil pH levels on non-irrigated paddocks have fluctuated between 5.9 and 6.5, while on irrigated paddocks, soil pH levels have ranged from 6.2-7.0.

The pH of the LNZ effluent is acidic (pH $\approx$ 4.5). The irrigation of acidic effluent, would be expected to decrease soil pH. It is important to note that the pH of the effluent has become acidic only recently. The increase in soil pH may be attributed to the years of irrigation of effluent with a pH of around 7.0 and/or the 1 tonne of lime applied per hectare per year since 1996. Information on the effect of acid pH effluent being irrigated onto pasture soils is scarce.

4.4.2 Carbon

Total carbon levels have increased slightly in all irrigated study paddocks at all depths (Table 4.10). Considering a significant decrease in bulk density of the irrigated paddocks there was a large increase in total carbon in the irrigated paddocks when the values are expressed on a weight basis. The largest increase in total carbon is at the 15-30 cm soil depth. The 14 year study paddocks have significantly greater ($P \leq 0.05$) total carbon at 7.5-15 and 15-30 cm soil depth than the non-irrigated study paddocks. The 14 and 40 year study paddocks have greater total carbon in the soil profile than the non-irrigated study paddocks. Below 30 cm, effluent irrigation has had little effect on total carbon levels.

Table 4.10 Total carbon for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (kg m^{-3})	9 years (kg m^{-3})	14 years (kg m^{-3})	30 years (kg m^{-3})	40 years (kg m^{-3})
0-7.5 cm	$82_a \pm 8.3$	$84_a \pm 0.1$	$88_a \pm 0.2$	$84_a \pm 9.2$	$93_a \pm 8.9$
7.5-15 cm	$56_a \pm 4.7$	$63_{ab} \pm 0.8$	$69_{cb} \pm 0.8$	$68_{cb} \pm 0.8$	$64_{ab} \pm 0.6$
15-30 cm	$27_a \pm 3.7$	$36_{ab} \pm 1.4$	$43_{cb} \pm 6.7$	$34_{ab} \pm 2.4$	$40_{cb} \pm 1.7$
30-45 cm	$15_a \pm 2.1$	$22_a \pm 7.1$	$20_a \pm 4.7$	$20_a \pm 2.2$	$21_a \pm 2.6$
45-60 cm	$13_a \pm 1.8$	$16_a \pm 3.3$	$13_a \pm 2.8$	$13_a \pm 0.6$	$15_a \pm 0.7$
Total (kg 0.6 m^{-3})	18.6	22.13	23.2	21.45	23.2

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

The increase in total C in the top 0-60 cm accounted for 7% (Refer to Appendix 1 for calculations) of the carbon applied with the effluent, based on annual loading rates and annual averages of composition over the last 9 years. Of the 93 % of applied C not accounted for in the soil, most of this was probably lost as CO_2 through decomposition (microbial respiration). Since a substantial portion of the carbon in the effluent is present as a soluble and fine particulate fraction, it is possible that some of this carbon would have been leached beyond 60 cm soil depth.

Easily oxidisable organic carbon values, as measured by dichromate oxidation, are very similar across all study paddocks at the 0-7.5 cm depth. The 14, 30 and 40 year study paddocks have significantly greater ($P \leq 0.05$) easily oxidisable organic carbon levels at the 7.5-15 cm soil depth. The largest increases in total carbon are found at the 7.5-15 and 15-30 cm soil depths. Below 30 cm, effluent irrigation has had little effect on organic carbon levels.

Table 4.11 Easily oxidisable organic carbon for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (kg m^{-3})	9 years (kg m^{-3})	14 years (kg m^{-3})	30 years (kg m^{-3})	40 years (kg m^{-3})
0-7.5 cm	$63_a \pm 3.9$	$61_a \pm 0.3$	$68_a \pm 0.0$	$62_a \pm 6.5$	$67_a \pm 4.0$
7.5-15 cm	$42_a \pm 3.2$	$48_{ab} \pm 2.2$	$55_b \pm 1.0$	$52_b \pm 0.3$	$50_b \pm 0.0$
15-30 cm	$21_a \pm 3.0$	$27_a \pm 0.9$	$29_a \pm 6.3$	$28_a \pm 2.7$	$29_a \pm 0.3$
30-45 cm	$13_a \pm 0.5$	$16_a \pm 5.6$	$15_a \pm 2.1$	$20_a \pm 10.2$	$18_a \pm 2.7$
45-60 cm	$11_a \pm 2.7$	$12_a \pm 3.1$	$10_a \pm 1.7$	$10_a \pm 0.7$	$12_a \pm 2.0$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

The greater increase of total carbon at depth in the irrigated soil, relative to non-irrigated soil, could be the result of redistribution of organic carbon from the surface layer through greater earthworm activity (Section 4.5.1) and/or leaching of organic carbon, when the soils were previously irrigated with alkaline effluent (Lieffering and McLay, 1995). The presence of greater earthworm biomass in the irrigated soils would promote carbon turnover and redistribution to depth in the soil. Application of effluent can alleviate seasonal moisture limitations in soils, thereby enabling longer periods of earthworm activity in irrigated soils compared with non-irrigated soils (Baker, 1998; Yeates, 1976; 1995).

It is conceivable that during irrigation with alkaline effluent, organic carbon was extracted, and remained soluble, irrespective of the pH of subsequent effluent applications, and continued to move slowly down the soil profile. The more permanent burrows formed by earthworms at the irrigated site would contribute to this process since earthworm burrows can conduct effluent to deeper depths in this soil type (Lieffering and McLay, 1995).

Effluent application also increases pasture production through increased nutrient supply and greater water availability during summer months (Hart and Speir, 1992; Cameron *et al.*, 1997). This would increase inputs of organic matter and possibly increase soil carbon.

4.4.3 Nitrogen

Total nitrogen levels have increased slightly in irrigated paddocks compared with non-irrigated paddocks, with the largest increases occurring in the 0-15 cm soil depth (Table 4.12). All irrigated study paddocks had greater ($P \leq 0.05$) total nitrogen levels at the 7.5-15 cm depth than the non-irrigated study paddocks. Below 30 cm, effluent irrigation has had no significant effect on total N levels in irrigated study paddocks.

Table 4.12 Total nitrogen for soils with varying years of effluent irrigation (mean ± standard deviation).

Soil Depth	Years of effluent irrigation				
	0 years (kg m^{-3})	9 years (kg m^{-3})	14 years (kg m^{-3})	30 years (kg m^{-3})	40 years (kg m^{-3})
0-7.5 cm	$8.01_a \pm 0.63$	$9.09_a \pm 0.15$	$8.89_a \pm 0.05$	$8.78_a \pm 0.84$	$9.83_a \pm 1.05$
7.5-15 cm	$5.43_a \pm 0.57$	$6.73_b \pm 0.06$	$6.66_b \pm 0.13$	$6.97_b \pm 0.19$	$6.85_b \pm 0.05$
15-30 cm	$2.32_a \pm 0.27$	$3.47_{ab} \pm 0.31$	$3.66_b \pm 0.70$	$3.37_{ab} \pm 0.24$	$4.15_b \pm 0.32$
30-45 cm	$1.18_a \pm 0.11$	$1.88_a \pm 0.66$	$1.36_a \pm 0.27$	$1.79_a \pm 0.19$	$1.93_a \pm 0.33$
45-60 cm	$0.94_a \pm 0.16$	$1.20_a \pm 0.20$	$0.79_a \pm 0.14$	$1.01_a \pm 0.00$	$1.24_a \pm 0.14$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

Degens *et al.* (2000) reported only a small increase in total nitrogen storage in the soil profile, where dairy factory effluent was applied to soil for 22 years. As reported in Table 4.3, nitrogen loading rates at the LNZ land treatment system are in excess of 450 kg N ha^{-1} $year^{-1}$. The results in Table 4.12 indicate not all the nitrogen irrigated in the effluent is accounted for in the soil. Therefore, nitrogen is likely to have been leached from the soil system as nitrate or lost as gaseous N through denitrification, especially under saturated conditions. This is reflected in the nitrate groundwater concentration under the irrigation farms (Figure 4.10).

Most of the nitrogen in the effluent is present in ammoniacal and organic N forms. Nitrogen in the applied effluent would be rapidly converted to nitrate in these relatively warm and moist soil conditions. Nitrate can then be rapidly lost from the topsoil through leaching and denitrification, especially under saturated conditions. Nitrate-N concentrations in the groundwater below the land treatment site are elevated; since monitoring began in 1991, median concentrations are above 13 g m^{-3} on both farms No. 1and No. 2 in comparison to < 1 g m^{-3} in the non-irrigated paddocks. In addition, the concurrent application of a readily available carbon source and nitrate would stimulate denitrification in these loamy soils. However denitrification losses from the system are unknown.

Soil samples from all study paddocks were analysed for nitrate-nitrogen and ammonium-nitrogen. Nitrate values measured were extremely variable (Tables 4.13 and 4.14). In general, both nitrate and ammonium levels have increased on irrigated paddocks compared with non-irrigated paddocks. The largest increases occur at the 7.5-15 cm soil depth (mean values of 28 $\mu g\ cm^{-3}$ and 8.75 $\mu g\ cm^{-3}$ on irrigated paddocks for nitrate and ammonium, respectively, compared with mean values of 9 $\mu g\ cm^{-3}$ and 4.84 $\mu g\ cm^{-3}$ on non-irrigated paddocks). There is considerable variation with nitrate values in relation to the number of years paddocks have been irrigated with effluent. Below 30 cm depth, on non-irrigated paddocks there is little NO_3^- compared with irrigated paddocks. Both the 14 and 40 year study paddocks have large nitrate concentrations below 30 cm, indicating NO_3^- is moving down the soil profile. There was no significant difference in the NH_4-N at lower depths between the effluent irrigated and non-irrigated paddocks suggesting that ammonium is not moving down the profile.

The effect of effluent irrigation on nitrate and ammonium was not consistent. This was expected as the quantity of NO_3-N and NH_4-N depends on a number of factors including climate and time of sampling.

Table 4.13 Nitrate-nitrogen values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (μg cm^{-3})	9 years (μg cm^{-3})	14 years (μg cm^{-3})	30 years (μg cm^{-3})	40 years (μg cm^{-3})
0-7.5 cm	$35_a \pm 24.1$	$20_a \pm 18.4$	$39_a \pm 12.6$	$67_a \pm 1.7$	$28_a \pm 4.5$
7.5-15 cm	$10_a \pm 9.2$	$14_a \pm 6.6$	$38_a \pm 18.4$	$39_a \pm 2.9$	$20_a \pm 0.2$
15-30 cm	$12_a \pm 15.3$	$23_a \pm 17.1$	$29_a \pm 0.6$	$38_a \pm 27.9$	$21_a \pm 6.6$
30-45 cm	$3_a \pm 0.91$	$35_a \pm 31.42$	$8_a \pm 0.96$	$14_a \pm 1.47$	$38_a \pm 15.16$
45-60 cm	$2_a \pm 0.41$	$7_a \pm 3.29$	$4_a \pm 2.37$	$7_a \pm 2.03$	$23_b \pm 12.92$

Values with a similar letter within a row are not significantly different at P≤0.05.

Table 4.14 Ammonia-nitrogen values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (μg cm^{-3})	9 years (μg cm^{-3})	14 years (μg cm^{-3})	30 years (μg cm^{-3})	40 years (μg cm^{-3})
0-7.5 cm	$7.8_a \pm 1.5$	$10.0_a \pm 3.3$	$10.1_a \pm 0.9$	$10.1_a \pm 1.2$	$9.2_a \pm 0.3$
7.5-15 cm	$4.8_a \pm 1.0$	$8.9_b \pm 1.3$	$9.1_b \pm 0.2$	$8.9_b \pm 0.1$	$8.2_b \pm 1.2$
15-30 cm	$5.8_a \pm 2.2$	$6.1_a \pm 1.6$	$8.2_a \pm 2.2$	$7.31_a \pm 0.3$	$7.2_a \pm 1.1$
30-45 cm	$5.0_a \pm 1.20$	$5.5_a \pm 0.4$	$6.5_a \pm 0.9$	$5.7_a \pm 0.3$	$5.6_a \pm 1.4$
45-60 cm	$5.7_a \pm 0.8$	$4.5_a \pm 0.9$	$6.2_{ab} \pm 0.3$	$5.2_a \pm 0.8$	$8.1_b \pm 0.0$

Values with a similar letter within a row are not significantly different at P≤0.05.

4.4.4 Cation exchange capacity and exchangeable cations

To a depth of 30 cm, cation exchange capacity values are slightly greater in irrigated study paddocks compared with non-irrigated paddocks (Table 4.15). The largest increases in CEC have occurred in the 14 and 30 year study paddocks at the 7.5-15 cm depth, where values are significantly greater (P≤0.05) than non-irrigated study paddocks. Below a depth of 30 cm, irrigation has had little impact on CEC values.

Table 4.15 Cation exchange capacity for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years ($meq\ cm^{-3}$)	9 years ($meq\ cm^{-3}$)	14 years ($meq\ cm^{-3}$)	30 years ($meq\ cm^{-3}$)	40 years ($meq\ cm^{-3}$)
0-7.5 cm	$0.30_a \pm 0.02$	$0.32_a \pm 0.05$	$0.32_a \pm 0.01$	$0.32_a \pm 0.02$	$0.32_a \pm 0.02$
7.5-15 cm	$0.21_a \pm 0.01$	$0.26_{ab} \pm 0.01$	$0.31_{bc} \pm 0.01$	$0.34_c \pm 0.01$	$0.28_{bc} \pm 0.03$
15-30 cm	$0.13_a \pm 0.02$	$0.14_a \pm 0.01$	$0.17_{ab} \pm 0.01$	$0.20_{bc} \pm 0.01$	$0.23_c \pm 0.01$
30-45 cm	$0.09_a \pm 0.02$	$0.11_a \pm 0.04$	$0.08_a \pm 0.01$	$0.12_a \pm 0.02$	$0.11_a \pm 0.02$
45-60 cm	$0.08_a \pm 0.00$	$0.09_a \pm 0.05$	$0.06_a \pm 0.01$	$0.08_a \pm 0.02$	$0.09_a \pm 0.02$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI (NZDRI, 1998) indicate CEC levels in irrigated paddocks are higher than non-irrigated paddocks when expressed on a weight basis. However, when converted to a volume basis, using the bulk density data in Table 4.5, the values NZDRI measured are very similar to values in Table 4.15.

An increase in soil organic matter content, resulting from the application of organic wastes can produce a related increase in soil cation exchange capacity (Bernal *et al.*, 1992; Stadelmann and Furrer, 1985). Regression analysis determined a strong relationship between cation exchange capacity and total soil carbon, at all depths sampled ($r^2 = 0.93$). Increase in soil pH could also be a reason for the increase in CEC in these soils containing variable charge components.

Exchangeable sodium levels have increased significantly ($P \leq 0.05$) on irrigated study paddocks compared with non-irrigated study paddocks at 0-7.5 cm depth (Table 4.16). The greatest increase in sodium levels was measured at the 7.5-15 cm depth, (7-9 fold). These increases are extremely large when compared with Keeley and Quin (1979) who reported a significant increase in sodium levels from 0.5 to 1.2 $cmol_c\ kg^{-1}$, with application of fellmongery effluent for 80 years. At 15-30 cm depth, only the 30 and 40 year study paddocks had sodium levels significantly greater ($P \leq 0.05$) than the non-irrigated study paddocks. There are no significant differences between any irrigated study paddocks at 0-15 cm depth. There is considerable variation in sodium levels with respect to the number of years paddocks have been irrigated. The 9 year study paddocks in particular, have relatively high sodium levels and the 14 year study paddocks have relatively low sodium levels, at all depths sampled. Below 30 cm, effluent irrigation has had little effect on sodium levels.

Table 4.16 Sodium values for soils with varying years of effluent irrigation (mean ± standard deviation)

	Years of effluent irrigation				
Soil Depth	0 years ($meq\ cm^{-3}$)	9 years ($meq\ cm^{-3}$)	14 years ($meq\ cm^{-3}$)	30 years ($meq\ cm^{-3}$)	40 years ($meq\ cm^{-3}$)
0-7.5 cm	$0.003_a \pm 0.0003$	$0.010_b \pm 0.0021$	$0.011_b \pm 0.0013$	$0.013_b \pm 0.0024$	$0.012_b \pm 0.0021$
7.5-15 cm	$0.002_a \pm 0.0010$	$0.015_b \pm 0.0036$	$0.013_b \pm 0.0001$	$0.017_b \pm 0.0001$	$0.020_b \pm 0.0021$
15-30 cm	$0.002_a \pm 0.0001$	$0.009_{abc} \pm 0.0001$	$0.005_{ab} \pm 0.0001$	$0.011_{bc} \pm 0.0055$	$0.015_c \pm 0.0024$
30-45 cm	$0.002_a \pm 0.0007$	$0.008_a \pm 0.0048$	$0.004_a \pm 0.0021$	$0.005_a \pm 0.0027$	$0.010_a \pm 0.0010$
45-60 cm	$0.002_a \pm 0.0001$	$0.007_a \pm 0.0042$	$0.003_a \pm 0.0003$	$0.005_a \pm 0.0015$	$0.006_a \pm 0.0001$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI (NZDRI, 1998) indicate sodium levels in irrigated paddocks are considerably higher than non-irrigated paddocks, in most cases a two-fold increase when expressed on a weight basis. However, when converted to a volume basis, the values NZDRI measured are very similar to values in Table 4.16. Since the start of the soil analyses, sodium values are generally decreasing on all farms with the odd fluctuation.

Exchangeable potassium levels have increased significantly ($P \leq 0.05$) on irrigated study paddocks compared with non-irrigated study paddocks at 0-45 cm depth (Table 4.17). The largest increases in potassium levels were measured at the 0-7.5 and 7.5-15 cm depth, (5-11 fold). These increases are exceptionally large when compared with the three-fold increase reported by Keeley and Quin (1979), after the irrigation of meatworks-fellmongery effluent for 80 years. Below 45 cm, effluent irrigation has had little effect on potassium levels. There are no significant differences between any irrigated study paddocks at any depth. There is considerable variation in potassium levels with respect to the number of years study paddocks have been irrigated. The 9 year study paddocks in particular, have relatively high potassium levels at all depths sampled.

Table 4.17 Potassium values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (meq cm^{-3})	9 years (meq cm^{-3})	14 years (meq cm^{-3})	30 years (meq cm^{-3})	40 years (meq cm^{-3})
0-7.5 cm	$0.005_a \pm 0.0027$	$0.027_b \pm 0.0131$	$0.023_b \pm 0.0038$	$0.031_b \pm 0.0017$	$0.027_b \pm 0.0033$
7.5-15 cm	$0.002_a \pm 0.0005$	$0.016_b \pm 0.0040$	$0.016_b \pm 0.0039$	$0.024_b \pm 0.0015$	$0.023_b \pm 0.0088$
15-30 cm	$0.001_a \pm 0.0009$	$0.010_b \pm 0.0021$	$0.010_b \pm 0.0030$	$0.012_b \pm 0.0016$	$0.012_b \pm 0.0004$
30-45 cm	$0.001_a \pm 0.0006$	$0.010_b \pm 0.0016$	$0.005_b \pm 0.0006$	$0.008_b \pm 0.0008$	$0.007_b \pm 0.0013$
45-60 cm	$0.001_a \pm 0.0002$	$0.010_b \pm 0.0047$	$0.004_a \pm 0.0009$	$0.007_a \pm 0.0017$	$0.006_a \pm 0.0009$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI (NZDRI, 1998) indicate potassium levels in irrigated paddocks are higher than non-irrigated paddocks, in most cases two-fold greater. Potassium levels are generally increasing from year to year; often with large annual increases and occasional sporadic fluctuations.

Calcium levels have increased significantly ($P \leq 0.05$) on irrigated study year paddocks compared with non-irrigated study year paddocks at 0-30 cm depth (Table 4.18). The greatest increase in calcium levels was measured at the 7.5-15 cm depth, three to four-fold increases. These increases are extremely large when compared with Keeley and Quin (1979) who reported a significant increase in calcium levels from 9.4 to15.4 $cmol_c\ kg^{-1}$ with application of fellmongery effluent for 80 years. However there is a considerable amount of Ca applied annually from liming of paddocks, as well as the Ca irrigated in the effluent. At the 30-45 cm sampling depth, only the 30 and 40 year study paddocks have significantly greater ($P \leq 0.05$) calcium levels than the non-irrigated study paddocks. There are no significant differences between any irrigated study paddocks at the 0-15 cm depth. Below 45 cm, effluent irrigation has had little impact on calcium levels of irrigated study paddocks. There is a general trend of increasing calcium levels with increasing years of effluent irrigation.

Table 4.18 Calcium values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (meq cm^{-3})	9 years (meq cm^{-3})	14 years (meq cm^{-3})	30 years (meq cm^{-3})	40 years (meq cm^{-3})
0-7.5 cm	$0.127_a \pm 0.038$	$0.205_b \pm 0.002$	$0.221_b \pm 0.018$	$0.216_b \pm 0.023$	$0.234_b \pm 0.003$
7.5-15 cm	$0.051_a \pm 0.009$	$0.171_b \pm 0.001$	$0.208_b \pm 0.019$	$0.215_b \pm 0.006$	$0.202_b \pm 0.014$
15-30 cm	$0.032_a \pm 0.011$	$0.073_b \pm 0.007$	$0.104_{bc} \pm 0.011$	$0.125_{cd} \pm 0.006$	$0.160_d \pm 0.011$
30-45 cm	$0.028_a \pm 0.005$	$0.052_{ab} \pm 0.019$	$0.040_{ac} \pm 0.008$	$0.066_{bc} \pm 0.009$	$0.081_b \pm 0.003$
45-60 cm	$0.029_a \pm 0.004$	$0.041_a \pm 0.014$	$0.026_a \pm 0.009$	$0.047_a \pm 0.013$	$0.056_a \pm 0.001$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI (NZDRI, 1998) indicate calcium levels in irrigated paddocks are higher than non-irrigated paddocks. Calcium levels are relatively consistent on farms No.1 and No.2, at around 0.12-0.13 meq cm^{-3} of soil and 0.16 meq cm^{-3} of soil respectively, and are generally increasing on farm No.3.

Magnesium levels are greater ($P \leq 0.05$) in irrigated study paddocks compared to non-irrigated study paddocks at the 7.5-30 cm depth (Table 4.19). The greatest increase in magnesium levels was measured in the 7.5-15 cm depth; a mean value of 0.027 meq cm^{-3} for irrigated study paddocks compared with 0.009 meq cm^{-3} for non-irrigated study paddocks. Below 45 cm, effluent irrigation has had little impact on magnesium levels of irrigated study paddocks.

Table 4.19 Magnesium values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (meq cm^{-3})	9 years (meq cm^{-3})	14 years (meq cm^{-3})	30 years (meq cm^{-3})	40 years (meq cm^{-3})
0-7.5 cm	$0.024_a \pm 0.008$	$0.028_a \pm 0.005$	$0.031_a \pm 0.001$	$0.030_a \pm 0.002$	$0.040_a \pm 0.021$
7.5-15 cm	$0.009_a \pm 0.003$	$0.025_b \pm 0.002$	$0.026_b \pm 0.001$	$0.031_b \pm 0.007$	$0.027_b \pm 0.002$
15-30 cm	$0.005_a \pm 0.001$	$0.016_b \pm 0.001$	$0.013_b \pm 0.001$	$0.018_b \pm 0.005$	$0.020_b \pm 0.001$
30-45 cm	$0.004_a \pm 0.001$	$0.009_{bc} \pm 0.000$	$0.006_{ab} \pm 0.000$	$0.009_{bc} \pm 0.001$	$0.011_c \pm 0.002$
45-60 cm	$0.004_a \pm 0.001$	$0.004_a \pm 0.001$	$0.004_a \pm 0.001$	$0.006_a \pm 0.001$	$0.008_a \pm 0.001$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI (NZDRI, 1998) indicate magnesium levels in irrigated paddocks are higher (approximately 0.022 meq cm^{-3} compared to 0.013 meq cm^{-3} in non-irrigated paddocks). Magnesium levels are generally increasing from year to year, often with large annual increases and occasionally sporadic fluctuations, especially on farm No.1.

Where effluent contains significant quantities of the exchangeable cations Ca^{2+}, Mg^{2+}, Na^+ and K^+, the base saturation of the soil may also be increased (Table 4.20). Base saturation has increased significantly ($P \leq 0.05$) in paddocks irrigated with effluent compared with non-irrigated paddocks, at 0-30 cm depth. Keeley and Quin (1979) reported similar increases at 0-7.5 cm depth, with a significant increase in base saturation from 66% to 87% with application of fellmongery effluent for 80 years. Following application of wool scour effluent for 7 years, Campbell *et al.* (1980) reported a significant increase in base saturation at two sampling depths (at 0-15 cm depth, from 78% in control soils to 103% in irrigated soils and at 15-30 cm, from 75% in control soils to 102% in irrigated soils). The largest increase in base saturation was measured at the 7.5-15 cm depth. Over all depths sampled, there is a general trend of increasing base saturation with increasing years of irrigation. The base saturation of the 40 year study paddocks is very high, even at 45-60 cm, indicating cations are moving down the profile and possibly being leached.

Table 4.20 Base saturation values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (%)	9 years (%)	14 years (%)	30 years (%)	40 years (%)
0-7.5 cm	$54_a \pm 9.5$	$86_b \pm 6.0$	$89_b \pm 0.5$	$91_b \pm 1.6$	$99_b \pm 1.3$
7.5-15 cm	$31_a \pm 3.5$	$87_b \pm 0.6$	$84_b \pm 3.0$	$85_b \pm 0.5$	$95_b \pm 1.8$
15-30 cm	$31_a \pm 5.7$	$79_b \pm 5.3$	$78_b \pm 9.2$	$82_b \pm 4.67$	$90_b \pm 6.7$
30-45 cm	$43_a \pm 13.5$	$70_{ab} \pm 3.4$	$75_{ab} \pm 10.6$	$77_b \pm 2.5$	$95_b \pm 6.6$
45-60 cm	$43_a \pm 4.4$	$71_{ab} \pm 12.1$	$60_{ab} \pm 6.7$	$85_b \pm 3.3$	$88_b \pm 16.8$

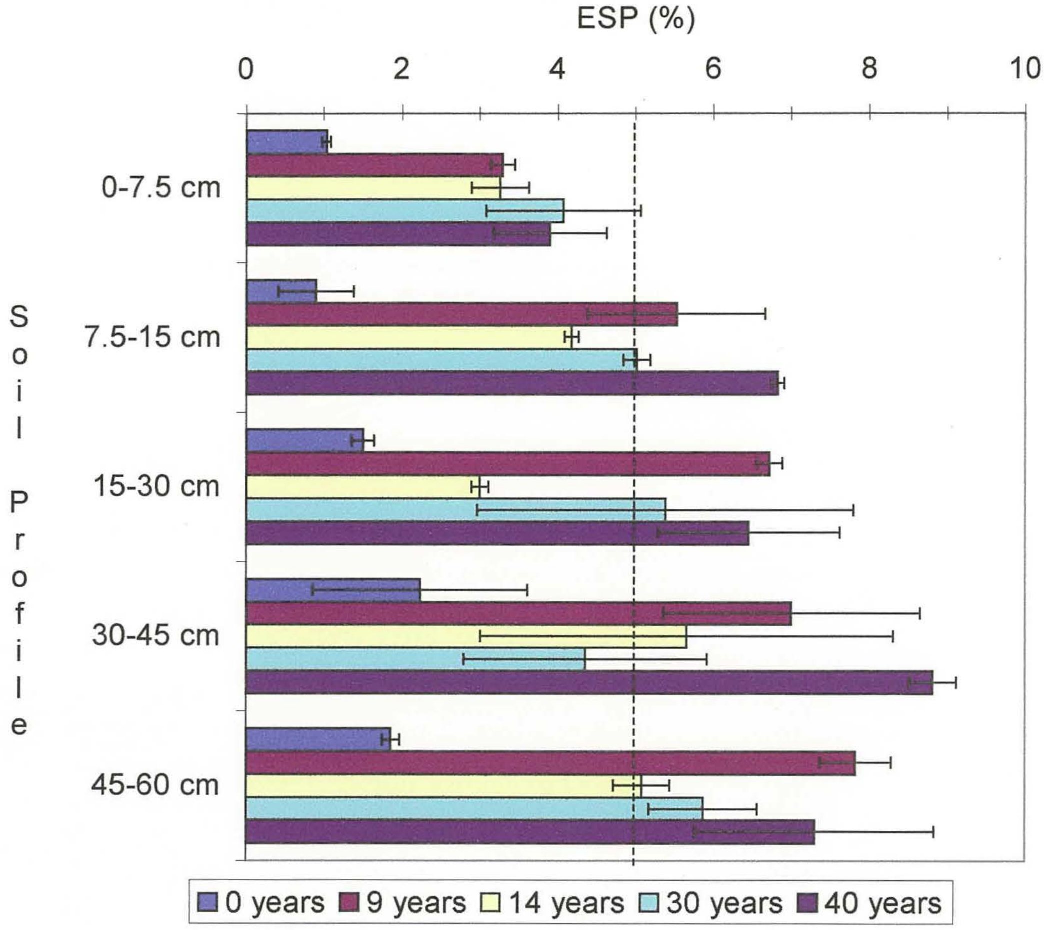
Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

The liming of irrigated paddocks at the LNZ irrigation farms will have an effect on base saturation levels. Liming gradually neutralises H^+ (and Al^{3+}) ions and increases the amount of calcium on exchange sites. As the H^+ ions are neutralised, the percentage base saturation increases and there is a gradual rise in soil pH. For all soils, as the base saturation decreases so does soil pH. However the exact relationship between base saturation and soil pH will vary between soils, depending on both the size of the exchange capacity and the types of exchange sites present (McLaren and Cameron, 1996). As soil pH and therefore OH^- ion concentration increases, the amount of negative charge on the colloid surfaces increases, resulting in greater retention of exchangeable basic cations.

4.4.5 Exchangeable Sodium Percentage

Exchangeable sodium percentage (ESP) is often used as an index of salinity problems in soils. The ESP values of irrigated study paddocks are significantly greater ($P \leq 0.05$) than non-irrigated study paddocks at 0-15 and 45-60 cm sampling depths (Figure 4.4). NZDRI reports also indicate ESP values are greater on irrigated paddocks and show some variation over time. There is a large increase in ESP values in the irrigated study paddocks between the 0-7.5 cm sampling interval and the 7.5-15 cm sampling interval. There is a general trend of increasing ESP with increasing years of effluent irrigation, except for the 9 year study paddocks, which have comparatively high ESP values, especially down the soil profile.

Figure 4.4 *Exchangeable sodium percentages for soil profiles with varying years of effluent irrigation*



NZDRI (1998) reported that ESP levels on farms No.1 and No. 2 were the lowest measured in recent years and ESP levels on all irrigated paddocks remained below the critical level of 5%. Traditionally, ESP values greater than 5% in the soils with crystalline layer silicate clay minerals are considered to promote deflocculation of soil with consequent loss of soil strength, soil structure and ponding of effluent occurring (NZDRI, 1993). However, the volcanic soils on the Lactose farms contain allophane (an amorphous clay mineral), which means they have inherently strong soil structure and soil strength. The strength of the soils is also strongly controlled by the moisture content of the soil (NZDRI, 1993). Figure 4.4 shows below 7.5 cm, many of the irrigated study paddocks have ESP values above 5%. The incidence of ESP values above 5% increases with increasing soil depth. NZDRI experience in the Waikato with the Horoitu series of soils (Similar to the Egmont series of soils), has shown that the ESP values on these soils can be as high as 15% on dry soils, without any deterioration in soil structure. NZDRI reported soil strength declined markedly when moisture exceeded 1.2 (v:v) and that deterioration in the soil structure will occur irrespective of the ESP. The dominant factor responsible for the loss of soil strength and soil structure on the Lactose farms is the moisture content of the soil (NZDRI, 1993)

4.4.6 Phosphorus

Olsen P values have increased significantly ($P \leq 0.05$) in paddocks irrigated with effluent compared with non-irrigated paddocks at 0-7.5 cm depth (Table 4.21) - a mean value of 267 $\mu g\ cm^{-3}$ for irrigated paddocks compared with 73 $\mu g\ cm^{-3}$ for non-irrigated paddocks. The largest increases in Olsen P were measured in the 30 and 40 year study paddocks at the 7.5-15 cm depth. This result is consistent with Degens *et al.* (2000) who reported an Olsen P value of 443 kg ha^{-1} in the 10-25 cm depth and at the 0-10 cm depth, a value of 366 kg ha^{-1}. Up to a depth of 30 cm, there is a general trend of increasing Olsen P with increasing years of irrigation. Irrigation has had little impact on Olsen P values of soil at a depth below 45 cm.

Table 4.21 Olsen P values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (μg cm^{-3})	9 years (μg cm^{-3})	14 years (μg cm^{-3})	30 years (μg cm^{-3})	40 years (μg cm^{-3})
0-7.5 cm	$73_a \pm 31$	$226_b \pm 36$	$251_b \pm 2$	$296_b \pm 25$	$295_b \pm 33$
7.5-15 cm	$30_a \pm 14$	$49_{ac} \pm 1$	$150_{bc} \pm 25$	$311_d \pm 48$	$350_d \pm 59$
15-30 cm	$14_a \pm 5$	$17_a \pm 3$	$33_a \pm 14$	$136_b \pm 2$	$168_b \pm 31$
30-45 cm	$10_a \pm 2$	$14_{ab} \pm 1$	$11_a \pm 2$	$41_c \pm 10$	$31_{bc} \pm 5$
45-60 cm	$10_a \pm 1$	$13_a \pm 1$	$10_a \pm 1$	$14_a \pm 3$	$13_a \pm 1$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI reports indicate that the Olsen P levels in irrigated paddocks are considerably higher than unirrigated paddocks. Their measurements show Olsen P levels generally increase or remain relatively constant from year to year although there have been some fluctuation reported on farm No. 1.

Regression analysis determined a strong relationship between Olsen P and years of effluent irrigation to a depth of 30 cm (r^2 = 0.71 at 0-7.5 cm, 0.96 at 7.5-15 cm and 0.94 at 15-30 cm). Trend analysis showed that effluent irrigation has resulted in an increase in Olsen P of 5.2 μg cm^{-3} per year at the 0-7.5 cm depth and 8.8 μg cm^{-3} per year at the 7.5-15 cm depth.

Soil samples from the 9 and 14 year study paddocks were analysed for total P (Table 4.22). Effluent irrigation has increased total P ($P \leq 0.05$) in the irrigated paddocks compared to non-irrigated paddocks at the 0-7.5 cm soil depth. The largest increases in total P were measured at the 0-7.5 cm soil depth. Below 15 cm, effluent irrigation has had little effect on total P levels.

Table 4.22 Total phosphorus values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation		
	0 years ($mg\ cm^{-3}$)	9 years ($mg\ cm^{-3}$)	14 years ($mg\ cm^{-3}$)
0-7.5 cm	$3.35_a \pm 0.48$	$5.01_b \pm 0.39$	$5.40_b \pm 0.05$
7.5-15 cm	$2.37_a \pm 0.33$	$3.42_a \pm 0.07$	$4.48_b \pm 0.36$
15-30 cm	1.60_a	n.a.	$1.63_a \pm 0.14$

n.a. Not analysed

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

Irrigation caused much greater increases in Olsen P than total P. This indicates that some of the P storage occurred in available forms that will benefit future pasture production on this site. However, there is a risk that continued accumulation of phosphorus in the irrigated soil may result in P levels toxic to plants, or induce deficiencies of other nutrients in the pasture (Marschner, 1986). Pasture levels of phosphorus at the irrigated site range between 0.45 and 0.50 %.

The increase in total P in the top 0-15 cm accounted for 52 % (Refer to Appendix 2 for calculations) of the P applied with the effluent, based on annual loading rates and annual averages of composition over the last 9 years. This storage can be attributed to the high P retention capacity of the predominantly allophanic clays in the soil. Of the 48 % of applied P not accounted for in the soil, most of this was probably leached or stored below 15 cm, since only a small percentage (≈2%) of the applied P was likely to be removed as milk.

4.4.7 Phosphate retention

Phosphate retention has decreased significantly ($P \leq 0.05$) on irrigated paddocks compared with non-irrigated paddocks at the 0-7.5 cm depth (Table 4.23); a mean of 44% for irrigated paddocks compared with 78% for non-irrigated paddocks. To a depth of 30 cm, there is a general trend of decreasing P retention, with increasing years of irrigation. This is most apparent at 0-7.5 cm depth. Below 30 cm, study paddocks have relatively consistent P retention values, apart from the 30 year study paddocks, which have relatively low P retention.

Table 4.23 Phosphate retention values for soils with varying years of effluent irrigation (mean ± standard deviation)

Soil Depth	Years of effluent irrigation				
	0 years (%)	9 years (%)	14 years (%)	30 years (%)	40 years (%)
0-7.5 cm	78_{a} ± 5.6	60_{b} ± 3.5	52_{bc} ± 1.4	37_{cd} ± 9.2	29_{d} ± 1.4
7.5-15 cm	86_{a} ± 4.5	81_{ab} ± 0.7	70_{b} ± 3.0	49_{c} ± 7.1	50_{c} ± 0.7
15-30 cm	91_{a} ± 2.5	90_{a} ± 0.0	87_{a} ± 1.4	71_{b} ± 2.8	76_{b} ± 3.5
30-45 cm	92_{a} ± 2.5	92_{a} ± 0.7	88_{a} ± 2.8	78_{b} ± 0.7	91_{a} ± 3.5
45-60 cm	90_{ab} ± 5.5	94_{a} ± 0.7	83_{ab} ± 4.9	79_{b} ± 0.7	93_{ac} ± 0.0

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

NZDRI reports indicate phosphate retention levels in irrigated paddocks are considerably lower than non-irrigated paddocks. These measurements show P retention levels generally decrease from year to year, although levels are fluctuating constantly between 45-60% on paddocks irrigated for 30 years on farm No. 1.

Regression analysis determined a strong negative relationship between phosphate retention and years of effluent irrigation (r^2 = 0.96 at 0-7.5, 0.94 at 7.5-15 and 0.81 at 15-30 cm depth). Trend analysis showed a decrease in P retention of 1.2% per year, at the 0-7.5 cm depth.

P retention measurement gives the potential maximum retention under the conditions presented in the method. It is important to note that P retention was measured at pH 4.6, which is not close to the pH measured under field conditions. Nevertheless a number of reasons could be attributed to the decrease in P retention with effluent application. These include, increase in soil pH, increase in organic C and increase in Olsen P levels. An increase in soil pH has often been shown to decrease the adsorption of P, which is attributed mainly to a decrease in the positive charge on the variable charge components. Similarly, an increase in organic carbon has been shown to block the adsorption sites and thereby decreases adsorption.

4.5 Phosphate adsorption isotherms

The phosphate adsorption isotherms were measured for soil samples from all study paddocks at 0-7.5 cm depth (Table 4.24). Phosphate adsorption isotherms were also determined for soil samples for the 40 year study paddocks at all depths. Phosphate adsorption isotherms were measured by equilibrating soil samples with different concentrations of phosphate solution. The native phosphate concentration in soil solution varied widely between the soils. In most soils, the concentrations of phosphate in the equilibrium solution after adsorption measurement were higher than the input concentrations, especially at lower levels of input concentration. This indicates that desorption of native phosphate has occurred at these concentrations. This created some difficulty in comparing the adsorption isotherms between the soils. So it was decided to interpolate the concentration of phosphate in the equilibrium solution at which there was no change in solution concentration after adsorption measurements. This is the equilibrium concentration at which neither adsorption nor desorption occurred and this concentration is known as 'null' concentration. The null concentration can be interpolated from the relationship between input concentration and equilibrium concentration (Figure 4.5). Above this concentration adsorption occurred and below which desorption occurred. An increase in 'null' concentration with effluent application indicates that the soils are susceptible to phosphate desorption.

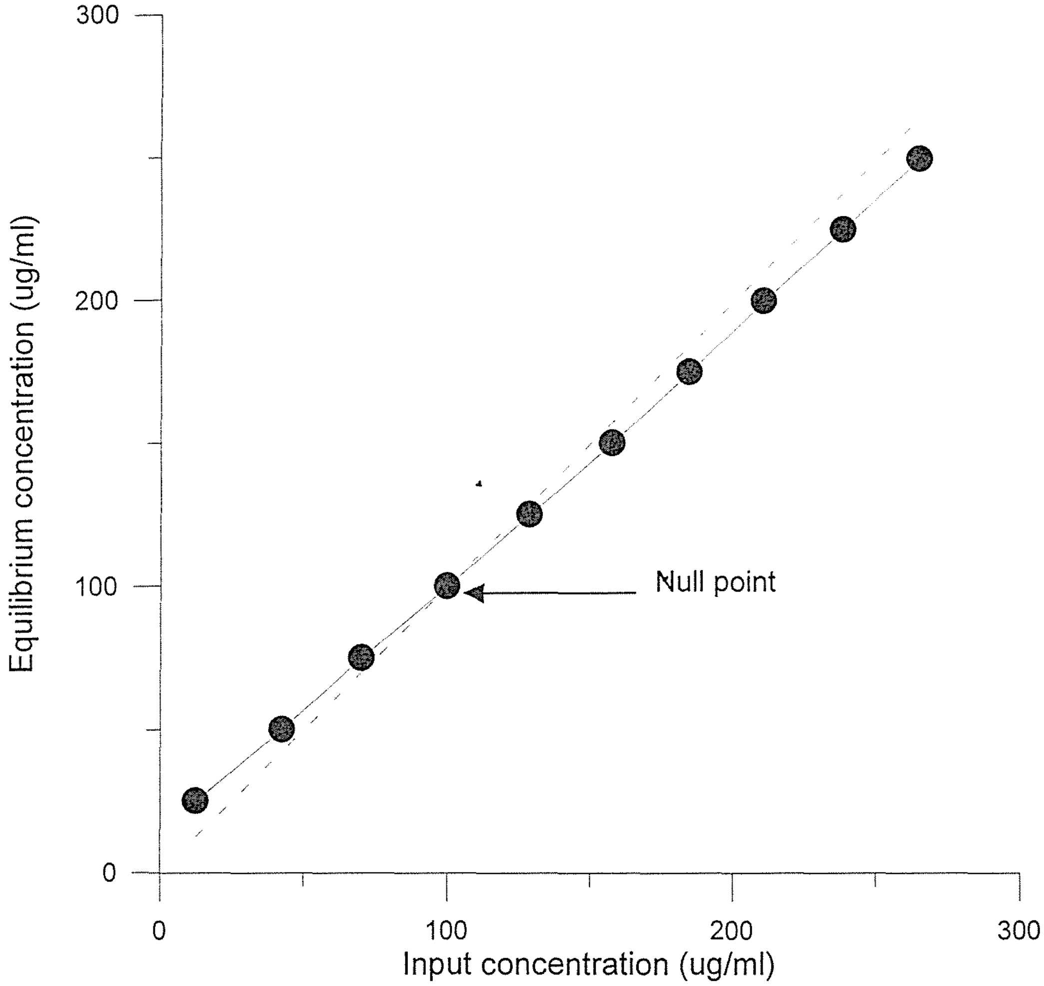


Figure 4.5 *Determination of null concentration for phosphate adsorption isotherms*

The dotted line shows 1:1 relationship and the solid line gives the measured relationship. The null point is the concentration at which these two lines intersect.

Table 4.24 Null concentration values for soils with varying years of effluent application

Paddock	Years of Irrigation	Null Concentration (μg ml^{-1})
F1 P5 0-7.5 cm	0	<1
F2 P2 0-7.5 cm	0	2.5
F3 P17 0-7.5 cm	0	<1
F3 P10 0-7.5 cm	9	15
F3 P24 0-7.5 cm	9	22
F1 P25 0-7.5 cm	14	22
F1 P32 0-7.5 cm	14	22
F1 P17 0-7.5 cm	30	103
F1 P18 0-7.5 cm	30	76
F2 P10 0-7.5 cm	40	111
F2 P16 0-7.5 cm	40	98
F2 P10 7.5-15 cm	40	50
F2 P16 7.5-15 cm	40	23
F2 P10 15-30 cm	40	6
F2 P16 15-30 cm	40	3
F2 P10 30-45 cm	40	<1
F2 P16 30-45 cm	40	<1
F2 P10 45-60 cm	40	<1
F2 P16 45-60 cm	40	<1

Effluent irrigation has increased the null point concentration of the irrigated paddocks from approximately 1-2 μg ml^{-1} up to approximately 100 μg ml^{-1} for paddocks irrigated for 30 and 40 years. There is a trend of increasing null point concentration with increasing period of effluent irrigation. This indicates that the concentration at which net adsorption occurred increased with increasing period of effluent irrigation (Figure 4.6). Phosphate desorption is likely to occur in the 0-15 cm of the paddocks irrigated for 40 years. However, below, 30 cm effluent irrigation has had little effect on null point concentration indicating these soils have the ability to adsorb additional phosphate.

Figure 4.6 *Relationship between Olsen P and null point concentration for soils with varying years of effluent irrigation*

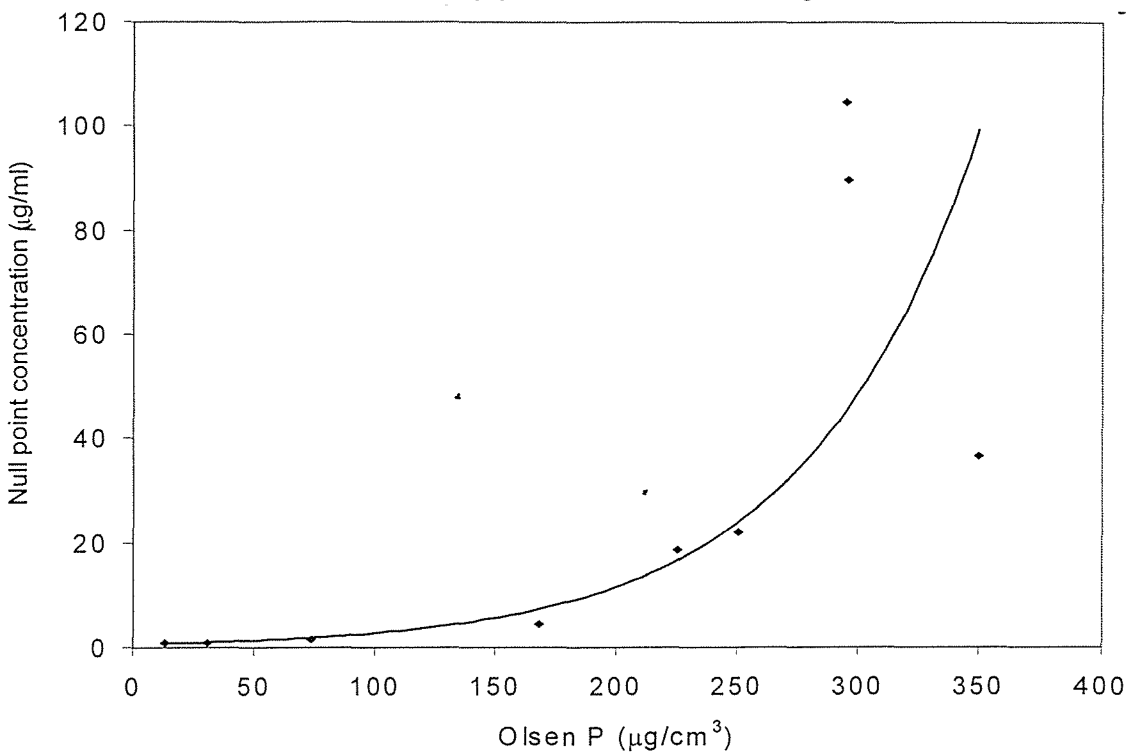


Figure 4.6 shows there is a strong exponential relationship (r^2=0.91) between Olsen P and the null point concentration of the LNZ soils. Above an Olsen P value of approximately 150 µg cm^{-3}, the null point concentration is increased dramatically. This indicates that P desorption is likely to occur around that Olsen P value. This is consistent with the relationship between Olsen P and soil solution P (Figure 4.7).

4.6 Soil solution phosphorus

Phosphorus concentrations in drainage waters from arable land range from <0.01 to occasionally above 1 mg $litre^{-1}$ (Culley *et al.*, 1983). Soil solution P has increased in the irrigated study paddocks compared to the non-irrigated study paddocks (Table 4.25). However, only the 40 year study paddock at the 0-7.5 cm depth and the 30 and 40 year study paddocks at the 7.5-15 cm depths, had significantly greater ($P \leq 0.05$) solution P values than the non-irrigated study year paddocks. The greatest increases and the largest solution P values were measured in the 7.5-15 cm range. There is a general trend of increasing solution P with increasing years of effluent irrigation over all depths sampled. Below 45 cm, effluent irrigation has had little effect on solution P.

Table 4.25 Solution P for soils with varying years of effluent irrigation (mean ± standard deviation)

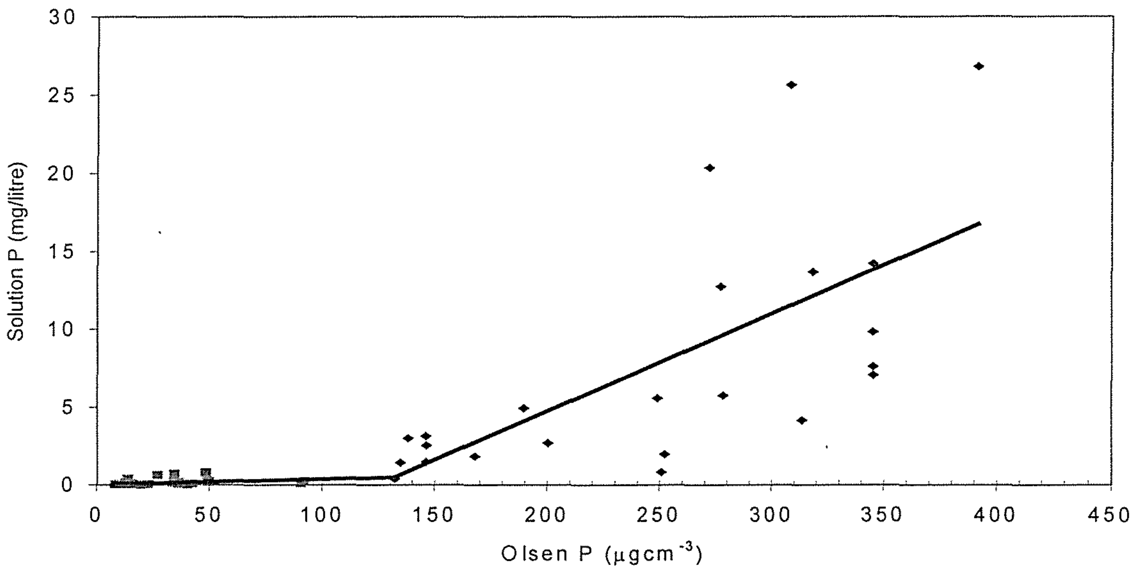
	Years of effluent irrigation				
Soil Depth	0 years (ppm)	9 years (ppm)	14 years (ppm)	30 years (ppm)	40 years (ppm)
0-7.5 cm	$0.15_a \pm 0.08$	$1.76_a \pm 0.93$	$3.77_a \pm 1.80$	$4.96_a \pm 0.80$	$17.01_b \pm 3.33$
7.5-15 cm	$0.08_a \pm 0.03$	$0.21_a \pm 0.03$	$1.12_a \pm 0.73$	$13.47_b \pm 0.73$	$26.22_c \pm 0.57$
15-30 cm	0.01_a	0.04_a	$0.08_a \pm 0.02$	$2.20_a \pm 0.77$	$5.21_a \pm 1.07$
30-45 cm	$0.03_a \pm 0.02$	n.s.	$0.02_a \pm 0.00$	$0.46_a \pm 0.33$	$0.68_a \pm 0.04$
45-60 cm	$0.01_a \pm 0.01$	$0.02_a \pm 0.01$	$0.03_a \pm 0.02$	$0.06_a \pm 0.03$	$0.08_a \pm 0.00$

n.s. No sample

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

To determine the relationship between soil Olsen-P concentrations and phosphorus in soil solution, the two factors were plotted on a graph (Figure 4.7). Data from the soil solution extraction and suction cups were used. The concentrations of P in solution were small for paddocks with less than 100 μg Olsen-P cm^{-3} (Figure 4.7). There was then a rapid increase in solution P in soil solution from soils containing greater than 100 μg Olsen-P cm^{-3} (Figure 4.7). Using a nonlinear regression analysis based on the least square method, a simple split-line model described this relationship well and was preferred to exponential models. The split-line model divides a curve into two linear sections, with a change in slope occurring at the so-called change point. The Olsen-P change point was estimated to be approximately 130 μg cm^{-3}.

Figure 4.7 Breakthrough curve showing the relationship between solution P and Olsen P for soils with varying years of effluent irrigation



The scatter around the line was considerable, reflecting the effect of soil, pasture and hydrological variables on P movement down the soil profile.

The results are similar to those found at the Broadbalk experiment at Rothamsted. Soils from this experiment, which have received annually for more than 100 years varying amounts and forms of P, now contain a wide range of Olsen P concentrations in the plough layer. An Olsen P 'change point' of 60 $\mu g\ cm^{-3}$ was obtained in this study. It was found that up to 60 μg Olsen P cm^{-3} soil, P was retained strongly in the plough layer. Above this P losses in the drainage water were much more closely related to Olsen-P than commonly suggested (Heckrath *et al.*, 1995). The change point in Figure 4.7 is considerably higher than reported by Heckrath *et al.* (1995). This is likely to be due to the very high P retention values for the LNZ soils.

These findings imply that, up to the change point concentration of Olsen-P, essentially all the P was retained in the rootzone layer. However, beyond this point, P in soil solution was much more closely related to Olsen-P concentrations in the rootzone. In soils such as that at Lactose, water does not move uniformly through the soil. There are likely to be mobile and immobile categories of water within the soil matrix. Phosphate sorption will need to be rapid in the pores through which water moves, although slower sorption processes may occur where there is immobile water.

Because of the high P fixation capacity of soils, vertical movement of P through the soil profile is generally considered of little importance, unless soils become P saturated, especially following heavy manure applications. Generally it is assumed that P leaving the topsoil is readily sorbed in P deficient subsoil. Heckrath *et al.* (1995) found the Broadbalk subsoils had a high maximum buffer capacity for P, were clearly undersaturated and maintained a low equilibrium soil solution P concentration. Nevertheless, they failed to retain P on plots with concentrations exceeding 60 mg Olsen P kg^{-1}. Whereas P saturation is considered to be the factor controlling its mobility in acid sandy soils, no satisfactory model exists to describe P mobility in the fine-textured more alkaline soils used for agriculture around the world (Heckrath *et al.*, 1995).

4.7 Effect of effluent on soil biological properties

The irrigation of effluent has had a varied influence on some of the soil biological properties of the soils. Both earthworm numbers and soil respiration rates have shown varied responses to effluent, with respect to the number of years of effluent irrigation.

4.7.1 Earthworm populations

When comparing irrigated study paddocks with non-irrigated study paddocks earthworm populations did not vary significantly, 692 earthhworms m^{-2} and 757 earthworms m^{-2}, respectively (Table 4.26). However, earthworm populations across the irrigated study paddocks are extremely variable. The 30 year study paddocks had an average of 1 003 earthworms m^{-2} which was significantly greater ($P \leq 0.05$) than the other irrigated study paddocks (9 and 40 years). All irrigated study paddocks have greater earthworm biomass m^{-2} than non-irrigated paddocks, a mean value of 388 g m^{-2} compared with 249 g m^{-2}. However, only the 30 year study paddocks had significantly greater ($P \leq 0.05$) earthworm biomass m^{-2} than the non-irrigated study paddocks. The mean individual earthworm mass is greater for paddocks receiving effluent, 0.62 g $earthworm^{-1}$ compared to 0.34 g $earthworm^{-1}$ for non-irrigated paddocks. The 9 and 40 year study paddocks showed significantly greater ($P \leq 0.05$) mean individual earthworm mass than non-irrigated paddocks. Earthworms in the 40 year study paddocks were on average, two fold heavier than those in non-irrigated paddocks.

Table 4.26 Earthworm populations, biomass and mean individual mass for soils receiving varying years of effluent irrigation (mean ± standard deviation)

Study Year	Earthworm Numbers ($n\ m^{-2}$)	Earthworm Biomass ($g\ m^{-2}$)	Mean individual earthworm mass (g)
0 years	761_{ab} ± 186	249_{a} ± 59	0.34_{a} ± 0.09
9 years	588_{a} ± 234	336_{ab} ± 142	0.62_{bc} ± 0.21
30 years	1003_{b} ± 343	443_{b} ± 208	0.47_{ab} ± 0.18
40 years	484_{a} ± 209	370_{ab} ± 234	0.76_{c} ± 0.30
Irrigated - average	692 ± 186	388 ± 198	0.62 ± 0.26

Values with a similar letter within a column are not significantly different at $P \leq 0.05$.

The results in Table 4.26 are consistent with Degens *et al.* (2000) who reported, that although a site irrigated with dairy factory effluent had lower earthworm populations than the non-irrigated site (a mean of 139 m^{-2} vs. 172 m^{-2}), the irrigated site had more earthworm biomass (68 g m^{-2} vs. 43 g m^{-2}). The mean individual biomass of earthworms was greater under irrigated than non-irrigated pasture (0.85 vs. 0.51 g).

The increases in earthworm biomass and mean individual biomass reported in Table 4.26 are quite small when compared to other reported increases. A study at Massey University assessed the effect of dairy effluent treatment on earthworm populations (Yeates, 1976). Earthworm numbers and weight were doubled on the irrigated area. McAuliffe (1978) reported that earthworm numbers at two dairy factory land treatment sites were five to six-fold higher than the corresponding control sites. The relatively small increase reported in Table 4.26 may be due to the inherently high earthworm numbers in the Egmont soils.

All irrigated study paddocks had a large degree of variation in earthworm populations, earthworm biomass and individual mass as reflected in the high standard deviations (Table 4.26). Generally, the degree of variation increases with increasing years of irrigation. Such large variations are not uncommon, as high spatial variations in earthworm populations have been reported at many land treatment sites (McAuliffe, 1978). There is a high degree of natural variation in earthworm populations (Fraser *et al.*, 1999).

4.7.2 Soil respiration rates

Basal soil respiration rates measured were very similar for all study years (Table 4.27). The only significant difference ($P \leq 0.05$) was between the 40 year study paddocks and the 9 year study paddocks. The substrate use of microbial communities was similar for the irrigated and non-irrigated soil with mineralisation of substrates in the 14, 30 and 40 year study paddocks slightly greater than the non-irrigated soil.

Table 4.27 Basal soil respiration rates (mg CO_2-carbon h^{-1} 100 g $soil^{-1}$) and use of glucose substrate by microbial communities (mean ± standard deviation)

Test	0 Years	9 Years	14 Years	30 Years	40 Years
Basal respiration	$2.46_{ab} \pm 0.37$	$2.31_{b} \pm 0.23$	$2.51_{ab} \pm 0.31$	$2.55_{ab} \pm 0.21$	$3.06_{a} \pm 0.45$
Glucose added	$3.62_{a} \pm 0.65$	$3.22_{a} \pm 0.17$	$4.30_{a} \pm 0.39$	$4.04_{a} \pm 0.54$	$4.31_{a} \pm 0.73$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

The soil microbial community changes in response to effluent application. Most studies report an increase in soil microbial activity following effluent addition. For example, simulated whey effluent application to soil induced a marked increase in the respiration rate within a few hours of application (McAuliffe, 1978). Degens *et al.* (2000) reported basal respiration rates were substantially greater ($P \leq 0.0001$) under soils irrigated with dairy factory effluent (irrigated: 6.24 μg C g^{-1} soil h^{-1}; non-irrigated: 4.02 μg C g^{-1} soil h^{-1}). In their study, mineralisation of substrates by microbial communities in the irrigated soil was greater than in the non-irrigated soil; between 1.5 and 3.0 fold greater for most substrates. For lactose and glucose, however substrate use was 5.6 and 9 fold, respectively greater, in the irrigated soil.

Soils that have previously received effluent have been observed to have a quicker increase in respiration rates following whey addition (McAuliffe, 1978). Therefore, it is difficult to ascertain why there is no significant increase in either respiration rates or substrate use in the current study. However, there are findings that support these results. In a study where soil cores were irrigated with treated sewage effluent or tap water, differences between soils were found to be greater than differences between treatments for total microbial populations and respiratory activity (Cairns *et al.*, 1978). It is important to note that the microbial activity, as measured by respiration rates was generally high in these fertile soils rich in organic matter.

4.8 Effect of effluent on water quality

Whilst no surface or groundwater samples were collected as part of the study, the Taranaki Regional Council have both surface and groundwater monitoring programmes.

4.8.2 Effect of effluent on surface water quality

Sampling of shallow drainage channels situated on and adjacent to the property immediately to the south of farm No.1 effluent irrigation area was undertaken by the regional Council at three sites. The results of the monitoring are shown in Table 4.28. This monitoring was designed to assess (over a 24-month period) whether LNZ irrigation activities were impacting on the drainage capability of the down-gradient neighbouring property. The three sites were situated in a small unnamed tributary of the Kaupokonui River and are shown in Figure 4.8.

Table 4.28 Results of surface water drainage quality monitoring of an unnamed tributary of the Kaupokonui River, from September 1994 to April 1995

Site		I1		I2		I3	
Parameter	Unit	No. of Reps	Range	No. of Reps	Range	No. of Reps	Range
Temperature	^0C	2	10 - 10.8	4	8.22-15.7	8	10.7-20.9
pH		2	7.1 - 7.2	4	6.8 - 7.1	8	6.9 – 7.8
Conductivity @ 20^0C	mS m^{-1}	2	59.4-68.8	4	36.3-63.4	8	35.8-41.8
COD	gm^{-3}	2	19 - 20	4	10 – 50	8	9 – 28
Sodium	gm^{-3}	2	68 - 82	4	31 - 75	8	40 – 52

Minimal flow was recorded in the drain at the upper site (I1) during the monitoring period and throughout summer, no flow was present at the upper roadside drain site (I2). Moderate to relatively low COD and sodium levels were measured in this section of the tributary. An improvement in downstream water quality continued to be indicated by results from site I3, probably due to dilution (by seepage and other drainage flows), filtration and polishing by drainage channel vegetation. Drainage channel water quality did not appear to deteriorate significantly during the summer months, when dilution was minimal in the lower drain, with the exception of a rise in COD during late summer.

The results in the upper drain (when not dry) indicate some similarity with groundwater conditions at the impact bore situated nearby on farm No.1, consistent with some impact from effluent irrigation on farm No.1.

4.8.2 Effect of effluent on groundwater quality

A monitoring programme to assess the impact of effluent irrigation on groundwater was begun in the 1991/92 season. Seasonal sampling of shallow groundwater bores is conducted by LNZ. Two bores located on each of the three effluent irrigation farms are sampled, with one bore ('control') on each property sited upslope of the irrigation area and the other bore ('impact') downslope of the irrigation area (Figure 4.9). A summary of the groundwater quality data previously collected by the Taranaki Regional Council, is presented in Table 4.29.

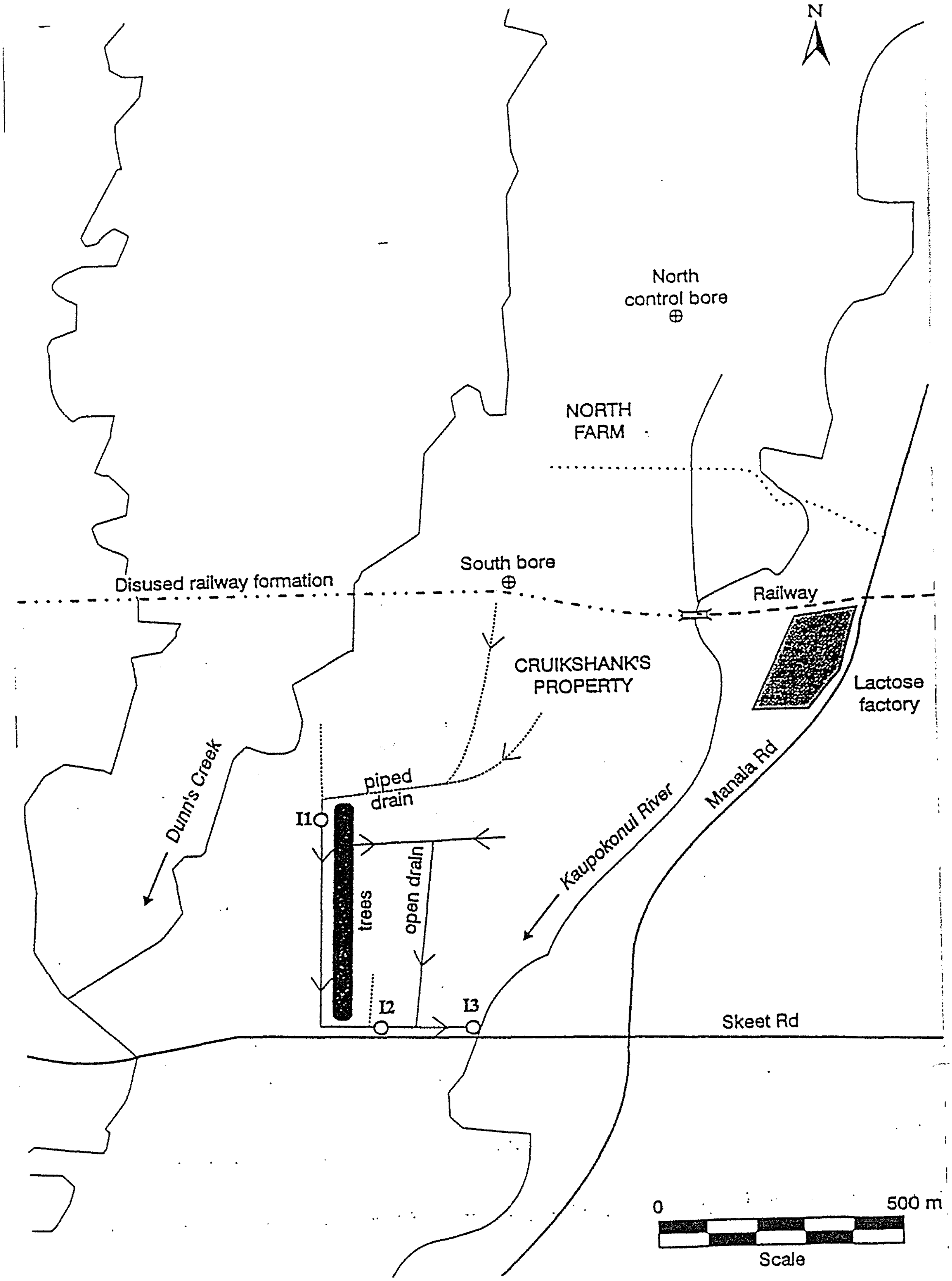


Figure 4.8 *Location of Farm No.1 irrigation surface water quality monitoring sites (I1 to I3) in unnamed tributary of the Kaupokonui River*

Table 4.29 Summary of previous TRC groundwater sampling performed during the period Oct 1991 to June 1999.

Parameter	pH			Conductivity		Sodium		Nitrate-N		COD (filtered)	
Unit				mSm^{-1}		gm^{-3}		gm^{-3}		gm^{-3}	
Farm Site	Bore	No	Range (median)	No	Range (median)	No	Range (median)	No	Range (median)	No	Range (median)
Farm No.1	Control	34	6.2 - 7.0 (6.5)	33	28 – 57 (30.2)	32	12 – 56 (25)	33	6.7 – 24 (8.85)	20	5 - 27 (8)
	Impact	33	6.1 - 7.8 (6.4)	32	42 – 82 (61)	31	50 – 179 (87.5)	32	2 – 32 (13.35)	18	5 – 42 (9)
Farm No.2	Control	26	5.9 – 6.9 (6.4)	26	60 – 149 (74.85)	26	79 – 136 (102	25	0.03 – 22 (0.56)	21	10 - 1600 (45)
	Impact 'original'	22	6.6 - 7.0 (6.7)	22	59 – 82 (71.3)	22	100- 157 (123)	22	3.8 – 29 (14.5)	17	5 - 35 (15)
	Impact 'new'	5	6.4 - 7.0 (6.7)	5	80 – 89 (87.4)	5	127– 153 (145)	5	0.8 - 6.0 (4.3)	5	6 - 34 (11)
Farm No.3	Control	24	6.4 – 6.9 (6.75)	25	21 – 37 (25.6)	24	28 – 35 (30.9)	24	0.01 - 3.4 (0.25)	21	4 – 21 (6)
	Impact 'original'	19	6.3 – 6.6 (6.4)	20	25 – 50 (36.3)	19	30 – 47 (36.3)	20	0.9 - 10.4 (6.7)	16	5 - 34 (9)
	Impact 'new'	4	6.4 – 6.8 (6.65)	5	40 – 65 (54)	5	59– 79 (73)	5	3.1 - 7.8 (6.4)	5	5 - 10 (8)

No. – number of samples

In 1996/97 the Regional Council undertook a groundwater contour and flow direction investigation. The report concluded the groundwater bores on farm No.1 were suitably orientated to detect effluent irrigation impacts. However, the groundwater bores on both farms No.2 and No.3 were not ideally orientated to detect the maximum effects on groundwater quality. Therefore, one monitoring bore ('impact') was installed on farm No.2 and farm No.3 in 1998. It is noted that the bores have been monitored for only a relatively short period of time, so additional accurate information is necessary before definite conclusions can be reached on the effects of effluent irrigation on groundwater quality.

From Table 4.29, it can be seen there are large variations in most parameters analysed. Some degree of variation is anticipated, due to stock management on the properties. Variations in the quality of the control bore's groundwater on farm No.2, particularly in conductivity and filtered COD levels is probably due to effluent irrigation in close proximity to this bore.

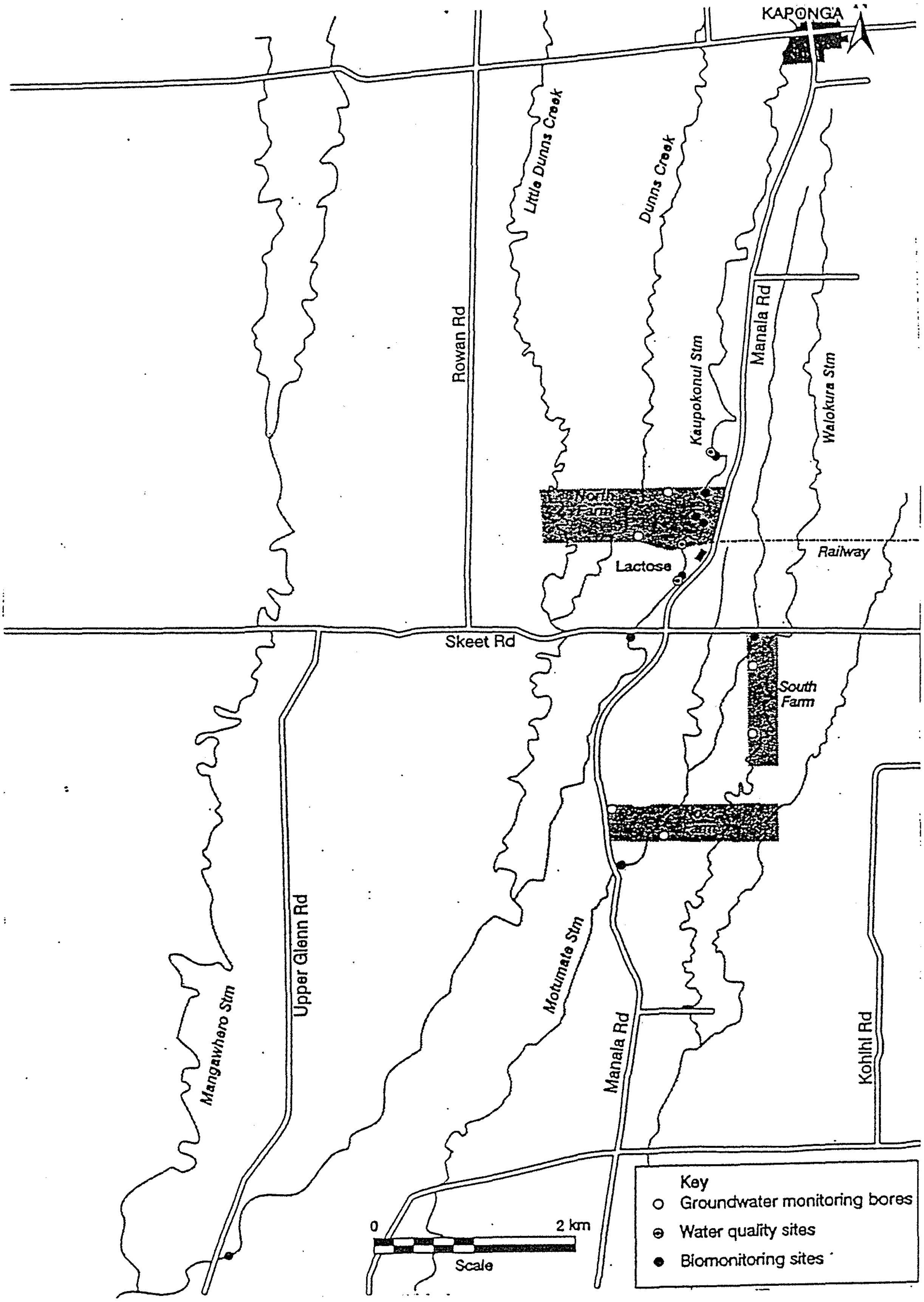


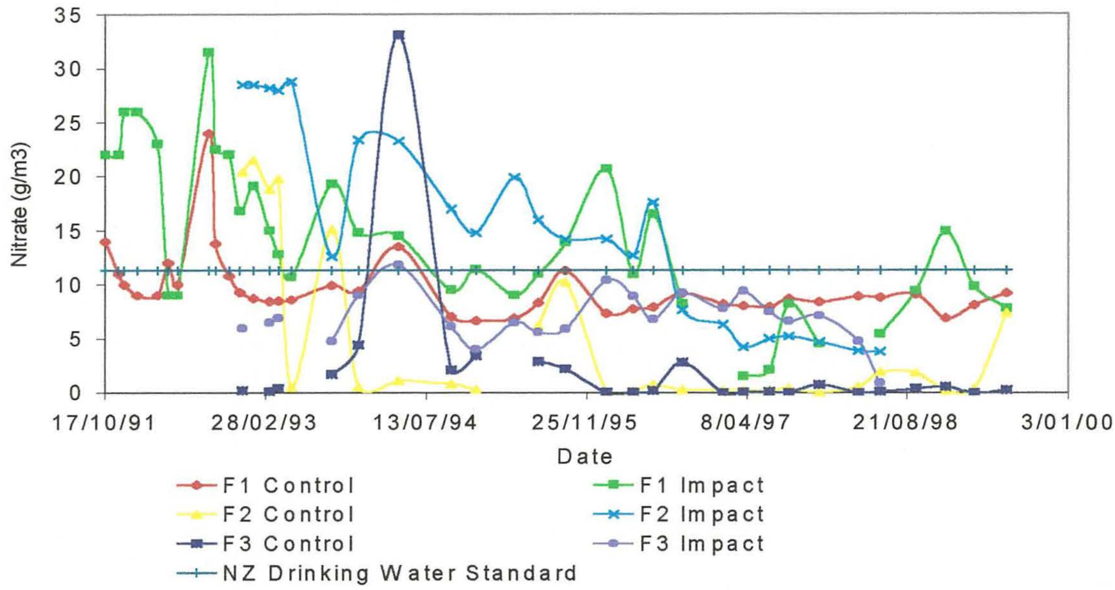
Figure 4.9 *Groundwater monitoring bores' locations on the three irrigation farms*

The extensive use of all three farms for spray irrigation of effluent has impacted on shallow groundwater to varying degrees, raising sodium, nitrate, conductivity and occasionally filtered COD levels (in the groundwater of most of the impact bores).

From the data in Table 4.29 and plotting each parameter on a time scale the following observations have been noted:

- Groundwater pH levels on farms No.1 and No.2 are decreasing slowly over time, while remaining reasonably constant on farm No.3. pH levels in the 'impact' bore on farm No.3 are consistently lower than the control bore (median value of 6.4 vs. 6.8).
- Groundwater conductivity and sodium levels are increasing in the 'impact' bores of all three farms, while levels remain reasonably constant in the 'control' bores on farms No.1 and No.3. The control bore on farm No.2 displays evidence of contamination and has been clearly affected by effluent irrigation, with conductivity levels measured as high as 149 mSm^{-1} and sodium concentrations up to 136 gm^{-3}.
- The irrigation of effluent onto farm No.1 has, on average, doubled the conductivity and resulted in a three-fold increase in sodium concentrations. In both bores located on farm No.2, groundwater sodium levels are the highest, on average 4 times higher than the No.1 farm control bore.
- Groundwater nitrate concentrations in both bores on all three farms are decreasing over time (Figure 4.10). Nitrate concentrations are greater in the 'impact' bores. The majority of the last several groundwater sampling events have measured nitrate concentrations below the New Zealand Drinking Water standard of 11.3 gm^{-3}.
- Groundwater COD levels are variable in all bores and further sampling and analysis need to be undertaken before any firm conclusions can be drawn.
- On farm No.3 some increases in contaminant levels are evident, however relative to the farm No.1 control bore the increase is insignificant.

Figure 4.10 *Nitrate-N concentrations in groundwater below Lactose irrigation farms*



The increase in conductivity indicates an increase in dissolved salts within the groundwater. The nitrate levels in certain periods, particularly before 1996, are of concern given the New Zealand Drinking Water standard of 11.3 gm^{-3} (as nitrate). The magnitude of the increase in sodium is of concern in terms of the effects of saline water on pasture and soil quality. There are no shallow groundwater users in the vicinity of the spray irrigation area, because of the usage of the Waimate West Rural Water Supply Scheme (TRC, 1999). However, there is concern because of both the immediate environmental effect on groundwater and because of the eventual discharge of shallow groundwater into local surface waterways and springs (TRC, 1999).

4.9 Grazing trial

A grazing trial was designed to assess the impact of grazing on recently irrigated pasture. A paddock irrigated for 40 years (paddock 7 farm No. 2) was selected for the trial, because it had consistently higher moisture contents than other irrigated paddocks. Since the bulk density of this treatment is very low, this site is likely to experience the greatest impact from grazing.

Paddock seven was fenced in half providing two treatments:

Treatment 1 (T1) - one irrigation run over half the paddock (34 millimetres of effluent was applied) with 12 hour grazing the following day.

Treatment 2 (T2) - two irrigation runs over the other half of the paddock (74 millimetres of effluent was applied) with 12 hour grazing the following day.

The initial soil moisture content of the soil in the grazing trial was unusually low due to the low rainfall in September (71mm compared with an average of 98) and October (36 mm compared with an average of 87 mm), preceding the trial.

4.9.1 Bulk density

Bulk density values for treatment 1 at both depths measured before and after the irrigation event were very similar (Table 4.30). Bulk density values for the 0-5 cm depth for treatment 2 were significantly lower ($P \leq 0.05$) after the irrigation event (78 g cm^{-3} vs 0.68 g cm^{-3}). This is a very unexpected result, as grazing is usually associated with compaction and therefore an increase in bulk density values. The result may be due to the spatial variability associated with measuring soil bulk density.

Table 4.30 Bulk density results from the grazing trial (mean ± standard deviation)

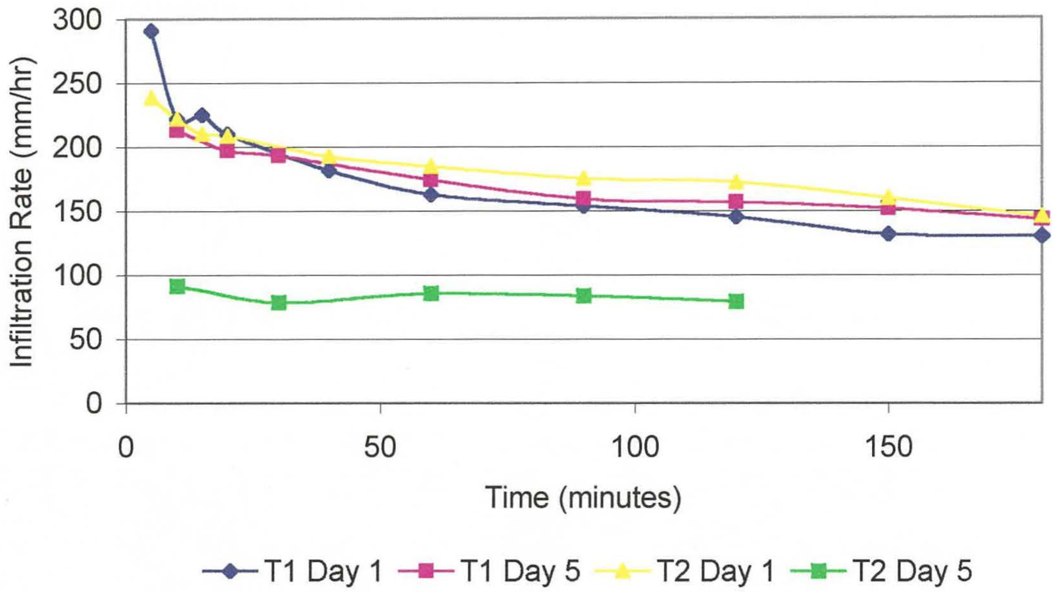
Treatment	Soil depth (cm)	Bulk Density (g cm^{-3})	
		Before Irrigation	After Irrigation
Treatment 1	0-5 cm	$0.66_a \pm 0.03$	$0.65_a \pm 0.03$
		$0.73_a \pm 0.04$	$0.72_a \pm 0.06$
Treatment 2	5-10 cm	$0.78_a \pm 0.04$	$0.68_b \pm 0.07$
		$0.79_a \pm 0.09$	$0.75_a \pm 0.07$

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

4.9.2 Soil Infiltration Rates

The infiltration rate for treatment 1 measured before and after irrigation were very similar, approximately 135 mm hr^{-1} (Figure 4.11). The infiltration rate for treatment 2 decreased significantly ($P \leq 0.05$) after the irrigation event, declining from approximately 145 mm hr^{-1} to 80 mm hr^{-1}, a 45% reduction.

Figure 4.11 *Soil infiltration rates before and after irrigation events*



Surface sealing as a result of animal grazing may have contributed to the decrease in soil infiltration rates. Bulk density measurements are unlikely to show any effect of surface sealing. There are a number of reports that document the reduction of soil infiltration following the irrigation of effluent. For example, McAuliffe (1984) found that soil permeability was reduced by 95% within 2 days of applying wool scour effluent. A decline in soil permeability has been reported at dairy factory effluent sites in New Zealand (Phillips, 1971; Parkin and Marshall, 1976; McAuliffe, 1978). In laboratory studies using undisturbed soil cores, McAuliffe *et al.* (1982) found that a single application of simulated whey effluent resulted in approximately a 50% decrease in soil permeability within two days. Repetitive doses of effluent lead to a decrease in soil permeability of over 99% in some cores. The decline was suggested to be a combination of both physical and biological blockage processes.

4.9.3 Surface Roughness

There is little difference between the chain measurements before and after grazing, with both treatments, indicating stock grazing did not cause any measurable surface roughness (Table 4.31). This is surprising considering the low bulk density values.

Table 4.31 Average chain measurements before and after grazing

Treatment	T1 Before	T1 After	T2 Before	T2 After
Average Length (cm)	99.3	97.0	99.4	99.1
Loss in length of chain (%)	2.3%		0.3%	
Roughness class	1		1	
Roughness name	Slightly rough		Slightly rough	

4.10 Bare patches

Bare patches have developed in some irrigated paddocks, particularly on farm No. 2. The bare patches range in size from approximately 5 m^2 to 20 m^2 and are characterised by soft, dark topsoil approximately 0-15 cm thick, overlying a considerably harder, lighter coloured layer. The bare patches have developed in low-lying areas where an excessive amount of effluent has ponded. In some instances, leaking hydrants have caused streams of effluent to flow down paddocks, resulting in bare patches. Vegetation does grow in the bare patches, however, it is very susceptible to grass pulling.

4.10.1 Bulk density

Bulk density values recorded at the 0-15 cm depth were very low ($P \leq 0.001$) compared with the 15-20 and 20-25 cm measurements, an average value of 0.47 g cm^{-3} over the four bare patches (Table 4.32). Bulk density values of the 40 year study paddocks at 0-7.5 cm and the bare patches in paddock 7 are very similar. However, the bulk density values measured in paddock 16, are considerably lower than those measured in the 40 year study paddocks (0.42 gcm^{-3} vs 0.52 gcm^{-3}). Bulk density values for the 15-20 and 20-25 cm depth are similar to values measured in the 40 year study paddocks (Table 4.5).

Table 4.32 Bulk density values for bare patches (mean ± standard deviation).

Soil Depth (cm)	Farm 2 Paddock 7 Bare Patch 1 (gcm^{-3})	Farm 2 Paddock 7 Bare Patch 2 (gcm^{-3})	Farm 2 Paddock 7 Bare Patch 1 (gcm^{-3})	Farm 2 Paddock 7 Bare Patch 1 (gcm^{-3})	Average (gcm^{-3})
0-15	$0.50_a \pm 0.02$	$0.55_a \pm 0.06$	$0.41_b \pm 0.03$	$0.42_b \pm 0.02$	0.47
15-20	$0.89_a \pm 0.03$	$0.97_b \pm 0.06$	$0.95_{ab} \pm 0.04$	$0.90_{ab} \pm 0.04$	0.93
20-25	$0.92_a \pm 0.04$	$0.90_a \pm 0.06$	$0.96_a \pm 0.03$	$0.93_a \pm 0.06$	0.93

Values with a similar letter within a row are not significantly different at $P \leq 0.05$.

4.10.2 Chemical analysis

The chemical analysis for the bare patches (Table 4.33) is compared with the chemical analysis for the 40 year study paddocks. Soil pH of the bare patches and the 40 year study paddocks is very similar, showing the same trend of increasing soil pH with depth. Total carbon values are lower in the bare patches, particularly in the topsoil (63 kg m^{-3} vs. 92 kg m^{-3}). Total nitrogen values are lower in the bare patches, particularly in the topsoil (6.9 kg m^{-3} vs. 9.83 kg m^{-3}). Olsen P values are considerably lower in the bare patches, half the value of the 40 year study paddocks. Phosphate retention values are slightly higher in the bare patches. Sulphur values are generally increasing with increasing soil depth. This shows that sulphur is mobile and is being leached through the soil profile.

Table 4.33 Chemical characteristics of the bare patches at 0-30 cm depth (mean ± standard deviation)

Soil Depth	pH	Total carbon (kg m^{-3})	Total nitrogen (kg m^{-3})	Olsen P (μg cm^{-3})	Phosphate retention (%)	Sulphur (μg S cm^{-3})
0-5 cm	6.8 ± 0.0	63 ± 7	6.9 ± 1.1	190 ± 39	38 ± 7	7.2 ± 3.3
5-10 cm	6.8 ± 0.0	50 ± 10	5.4± 1.4	171 ± 31	51 ± 3	8.6 ± 4.4
10-15 cm	6.9 ± 0.0	33 ± 9	3.6 ± 1.2	130 ± 29	63 ± 3	4.0 ± 0.2
15-20 cm	7.1 ± 0.0	38 ± 15	4.1 ± 1.8	161 ± 2	75 ± 7	14.2 ± 4.6
20-25 cm	7.1 ± 0.1	34 ± 9	3.8 ± 1.1	85 ± 4	82 ± 6	14.6 ± 4.4
25-30 cm	7.2 ± 0.0	33 ± 9	3.8 ± 1.1	33 ± 13	86 ± 5	17.1 ± 0.9

CEC values are slightly lower on the bare patches (Table 4.34) compared with the 40 year study paddocks. Sodium values are very similar in the topsoil but below this the sodium values are notably lower in the bare patches. Potassium levels measured in the bare patches were lower than the 40 year study paddocks, in the 0-15 cm range, but higher in the 15-30 cm range. Calcium and magnesium levels are generally lower in the bare patches throughout the profile. All chemical soil measurements are lower in the bare patches compared with paddocks irrigated for 40 years. It was anticipated the values would be higher, due to the large volumes of effluent the bare patches have received. However, erosion of the surface soil in the bare patches by surface runoff may have may have stripped the most fertile soil.

Table 4.34 Cation exchange capacity of the bare patches at 0-30 cm depth (mean ± standard deviation)

Soil Depth	K (meq cm^{-3})	Ca (meq cm^{-3})	Mg (meq cm^{-3})	Na (meq cm^{-3})	CEC (meq cm^{-3})
0-5 cm	0.019 ± 0.001	0.185 ± 0.034	0.023 ± 0.001	0.013 ± 0.002	0.24 ± 0.04
5-10 cm	0.017 ± 0.004	0.141 ± 0.034	0.019 ± 0.004	0.008 ± 0.001	0.20 ± 0.04
10-15 cm	0.017 ± 0.007	0.131 ± 0.054	0.012 ± 0.001	0.006 ± 0.001	0.19 ± 0.08
15-20 cm	0.017 ± 0.004	0.135 ± 0.033	0.015 ± 0.003	0.010 ± 0.001	0.22 ± 0.03
20-25 cm	0.018 ± 0.002	0.131 ± 0.005	0.012 ± 0.003	0.009 ± 0.000	0.18 ± 0.02
25-30 cm	0.016 ± 0.002	0.101 ± 0.002	0.016 ± 0.007	0.009 ± 0.001	0.15 ± 0.02

4.11 Effect of effluent irrigation on pasture

Pasture samples were collected in February 1999 and analysed for nutrient status by AgResearch. The results indicate that there is little variation between the irrigated paddocks on the three farms (Table 4.35). The majority of the results are in the normal range recommended by AgResearch or slightly above the range. Molybdenum levels are above 1 ppm, which may induce copper deficiency. However, copper levels on the LNZ irrigation farms are twice the recommended range.

Table 4.35 Nutrient status of pasture with varying years of effluent irrigation

Element	F1 P12 (30 yrs)	F1 P10 (2 yrs)	F3 P10 (9 yrs)	F3 P7 (9 yrs)	F2 P22 (40 yrs)	F2 P16 (2 yrs)	Normal Range
Nitrogen	4.66 %	4.51 %	4.01 %	4.46 %	4.93 %	4.64 %	4.50-5.50 %
Phosphorus	0.45 %	0.50 %	0.46 %	0.39 %	0.48 %	0.51 %	0.35-0.40 %
Sulphur	0.45 %	0.37 %	0.34 %	0.34 %	0.34 %	0.34 %	0.28-0.40 %
Magnesium	0.29 %	0.23 %	0.23 %	0.31 %	0.24 %	0.25 %	0.16-0.22 %
Calcium	0.55 %	0.55 %	0.43 %	0.47 %	0.51 %	0.55 %	0.25-0.50 %
Sodium	0.32 %	0.39 %	0.28 %	0.25 %	0.38 %	0.34 %	-
Potassium	3.68 %	3.42 %	3.72 %	4.21 %	3.48 %	3.46 %	3.00-3.50 %
Manganese	33 ppm	35 ppm	62 ppm	34 ppm	40 ppm	39 ppm	25 - 30 ppm
Zinc	43 ppm	40 ppm	87 ppm	34 ppm	38 ppm	44 ppm	20 - 50 ppm
Copper	14 ppm	14 ppm	13 ppm	14 ppm	14 ppm	15 ppm	6 - 7 ppm
Iron	155[2] ppm	334[3] ppm	135[4] ppm	115[4] ppm	375[3] ppm	209[2] ppm	50 - 60 ppm
Cobalt	0.05 ppm	0.10 ppm	0.06 ppm	0.02 ppm	0.11 ppm	0.05 ppm	-
Molybdenum[1]	3.73 ppm	3.99 ppm	4.63 ppm	4.04 ppm	3.34 ppm	3.53 ppm	-
Selenium	0.11 ppm	0.04 ppm	0.03 ppm	0.01 ppm	0.03 ppm	0.06 ppm	-

Animal Health Maximum Requirements

[1] Molybdenum levels greater than 1.00 ppm could induce copper deficiency

Effect of Soil Contamination

[2] Iron concentration indicates that the pasture sample was slightly contaminated with soil, which may elevate Cobalt and Selenium values to a small extent.

[3] Iron concentration indicates that the pasture sample was contaminated with soil, which may elevate Cobalt and Selenium values to a significant extent.

[4] Iron concentration indicates that the pasture sample was free of soil contamination.

The study paddocks have similar values for most pasture quality parameters analysed (Table 4.36). Soluble sugars and starch is higher on the non-irrigated study paddocks (a mean value of 5.7 g 100 g^{-1} DM) than irrigated study paddocks (a mean value of 3.8 g 100 g^{-1} DM). There is also a considerable difference between the dietary cation-anion difference (DCAD) of the irrigated and non-irrigated study paddocks. Non-irrigated paddocks had a mean value of 577 meq kg^{-1} DM compared with 749 meq kg^{-1} DM for the irrigated study paddocks. This is a reflection of the large amounts of potassium and sodium being irrigated in the effluent and the increase in base saturation values of the irrigated study paddocks.

Table 4.36 Near-infrared reflective spectrometry predicted results for pasture receiving varying years of effluent irrigation

	Units	0 years	9 years	14 years	30 years	40 years
Dry matter	g $100g^{-1}$	96.4	94.7	95.8	95.5	95.5
Crude protein	g $100g^{-1}$ DM	20.1	21.0	22.8	22.1	21.3
Lipid	g $100g^{-1}$ DM	4.2	3.7	4.3	3.9	4.1
Ash	g $100g^{-1}$ DM	11.0	13.6	12.7	13.7	13.7
Fibre – Acid detergent fibre	g $100g^{-1}$ DM	30.2	32.3	30.7	31.9	32.0
- Neutral detergent fibre	g $100g^{-1}$ DM	54.3	55.8	54.1	52.3	53.2
Soluble sugars and starch	g $100g^{-1}$ DM	5.7	3.6	3.7	4.1	3.9
Dietary cation-anion difference	meq kg^{-1} DM	577	756	730	761	748
Organic matter digestibility	g $100g^{-1}$ DM	73.2	74.7	77.1	78.2	76.8
Estimated metabolisable energy	MJ kg^{-1} DM	10.4	10.4	10.8	10.8	10.6

It has been observed that DCAD values for most pastures in New Zealand range from 200 to 800 meq kg^{-1} DM, indicating cation surplus for anions (Bolan *et al.*, 2000). This normally results from the luxury uptake of potassium by pasture resulting in high concentration of potassium in herbage. Dietary cation-anion balance is important because animals attempt to maintain systemic acid-base balance and osmotic pressure in order to protect the integrity of cells and membranes and to optimise biochemical and physiological processes (Wilson, 1999). When herbage with excess cations over anions (positive DCAD) is fed to animals the concentration of alkali ions such as bicarbonate ions increases in body fluids resulting in alkalosis. There have been conflicting reports on the optimum levels of DCAD values required for dairy cattle (Roche, 1997). It has however been shown that diets with high DCAD values tend to increase the incidence of milk fever and supplementation of pre-calving rations with anionic salts (low DCAD values) reduce the incidence of milk fever (Beede, 1992; Wilson, 1996).

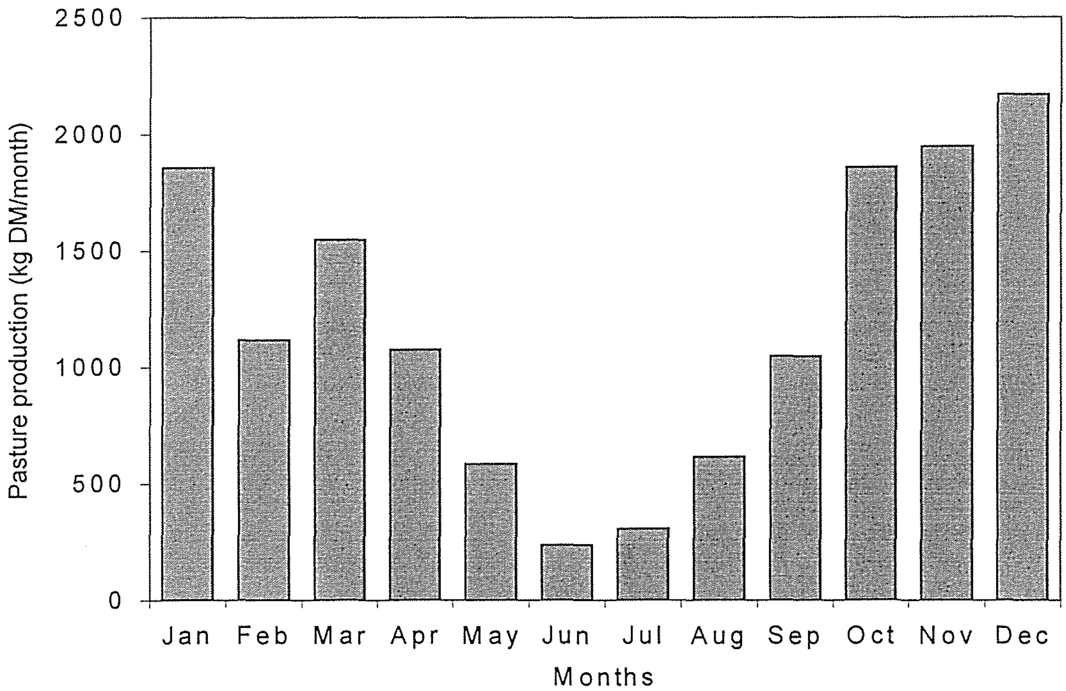
The Grass Stagger Index (GSI) was calculated using the data in Table 4.36. The index was calculated using the equation:

$$\text{GSI (meq kg}^{-1}\text{ DM)} = \text{K/(Ca} \pm \text{Mg)}.$$

The GSI values ranged from 4.33 to 5.63. These values are high when compared to those reported by Bolan *et al.* (2000). GSI values of 1.35–3.46 were reported following the addition of dairy effluent at varying rates. It has been suggested that when GSI values in the pasture are in excess of 2.2 the risk of hypomagnesaemia (grass staggers) development is enhanced (Bolan *et al.*, 2000). This condition is generally associated with animal serum magnesium levels less than 1.0-1.5 mg 100 ml^{-1}, compared to normal levels in cattle of 1.7-3.0 mg 100 ml^{-1}. This arises due to diets low in magnesium and additionally potassium induced inhibition of magnesium absorption in the rumen.

Pasture dry matter production figures were supplied by the farm supervisor. The annual average dry matter production is approximately 14 400 kg ha^{-1} (Figure 4.12).

Figure 4.12 *Annual pasture dry matter production for Lactose irrigation farms*



On one occasion during November, pasture burning was noted in large areas of paddocks on farm No. 2. Irrigating effluent that was too hot caused the pasture burning. The problem was resolved within the factory and no pasture burning was observed after this.

Pasture pulling was noted in a large number of irrigated paddocks. Pasture pulling (where grazing animals uproot plants) is a regular problem at grazed land treatment sites. Thomas and Evans (1975) noted that pasture pulling generally occurred on free draining soil, particularly after heavy rain following a dry spell. Pasture pulling was reported as being more extensive in shallow-rooted plant species, particularly if they were growing in a highly fertile soil. Pasture pulling at whey application sites was reported by McAuliffe (1978). He suggested that this was probably the result of loss of soil strength, due to high moisture content and a tendency for plants to only develop shallow root systems under effluent irrigation.

4.12 Historical sacrifice area

Historically, paddock 14 on farm No.2 has been subjected to excessive loadings of effluent. The paddock was very wet, and often suffered from ponding and pugging problems, as well as weed infestation and bare patches. It was thought analyses of soil samples from paddock 14 might give an indication of the nutrient levels that can be expected in other irrigated paddocks, at some point in the future. However, the nutrient levels for paddock 14 (Tables 4.37 and 4.38) are somewhat lower than those reported for the 40 year study paddocks.

Soil pH values are very similar between paddock 14, (farm No.2) and the 40 year study paddocks. Total carbon and total nitrogen values are higher in the 40 year study paddocks at the 0-7.5 cm depth and similar at all other depths sampled. Olsen P values are lower in paddock 14 than the 40 year study paddocks at all depth sampled.

Table 4.37 Chemical characteristics of paddock 14, farm No. 2

Soil Depth	pH	Total carbon ($kg\ m^{-3}$)	Total nitrogen ($kg\ m^{-3}$)	Olsen P ($\mu g\ cm^{-3}$)	Phosphate retention (%)	Sulphur ($\mu g\ S\ cm^{-3}$)
0-7.5 cm	6.7	70	7.4	225	47	7
7.5-15 cm	6.7	71	7.6	266	54	7
15-30 cm	7.0	39	4.2	137	80	8
30-45 cm	7.2	23	2.4	22	92	22
45-60 cm	7.2	15	1.5	11	94	57

CEC values are similar or slightly lower in paddock 14, farm No.2, compared with the 40 year study paddocks. Sodium levels are slightly higher in the 40 year study paddocks at the 7.5-30 cm depth. Potassium levels are similar between the paddocks. Calcium levels measured are slightly greater in paddock 14, at 7.5-60 cm depth. Magnesium levels are greater at the 0-7.5 cm depth in the 40 year study paddocks and then similar with paddock 14 down the soil profile.

Table 4.38 Cation exchange capacity of paddock 14, farm No. 2.

Soil Depth	K (meq cm^{-3})	Ca (meq cm^{-3})	Mg (meq cm^{-3})	Na (meq cm^{-3})	CEC (meq cm^{-3})
0-7.5 cm	0.024	0.202	0.028	0.012	0.273
7.5-15 cm	0.020	0.236	0.034	0.011	0.319
15-30 cm	0.014	0.156	0.018	0.008	0.175
30-45 cm	0.010	0.097	0.009	0.007	0.122
45-60 cm	0.008	0.077	0.007	0.005	0.096

CHAPTER 5

The future of the land treatment system

5.1 Introduction

Due to the number of dynamic factors involved in the operation, it is very difficult to foresee with any degree of certainty the future of a land treatment system. This is particularly true for the Lactose New Zealand system, due to its unique and variable effluent composition, soil type, climatic conditions and land use patterns. This point notwithstanding, the present study has identified some trends which can be used to estimate the rate of change in soil properties due to the addition of effluent under the present conditions. This is important to ensure the sustainable management of LNZ effluent.

5.2 Soil physical factors

Effluent irrigation has had a significant impact on some of the soil physical properties on the LNZ irrigation farms, in particular, the large reduction in bulk density.

Bulk density values on paddocks irrigated for 40 years have decreased to approximately 0.5 g cm^{-3}. Trendline analysis shows bulk density values on average, decrease 0.009 g cm^{-3} per year on irrigated paddocks at 0-5 cm depth. There is no indication bulk density values will not continue to decline. Bulk density values on the bare patches in paddock 16 (sites of major application due to breakdown of irrigators and flooding of effluent), farm No.2, were measured at approximately 0.4 g cm^{-3}. There is considerable difference in the 'softness' of the bare patches and paddocks irrigated for 40 years. There is the risk of further decrease in soil bulk density, combined with increased soil moisture content, resulting in treading damage by grazing animals. While the grazing trial indicated grazing animals had little impact on soil physical measurements, soil moisture contents during the trial were relatively low. During the periods in which the soil moisture is high, treading damage is likely to occur. The practice of grazing all stock off the farms during winter should continue. This will not only avoid possible treading damage but allow previous damage to repair.

As mentioned in the previous chapter, all soil chemical and some soil physical data is presented on a volume basis. This is due to the large change in soil bulk density at the 0-5 cm soil depth, from 0.89 gcm^{-3} in non-irrigated paddocks to 0.52 gcm^{-3} in paddocks irrigated for 40 years. Due to the large differences in soil density, it is more accurate to present data such as organic carbon content and soil moisture on a volume basis. However, to do so, masks some differences that may be developing between irrigated and non-irrigated paddocks.

Soil penetrometer measurements showed that measurements of soil strength on the irrigated paddocks was significantly lower ($P \leq 0.0001$) than the non-irrigated paddocks. This can be attributed mainly to the large difference in bulk density between irrigated and non-irrigated paddocks. It is very difficult to remedy this under the current land treatment system. Removal of stock during wet conditions will reduce the impact of treading damage.

Effluent irrigation appears to have had little effect on soil infiltration rates. Shallow subsoiling of the soil, removal of stock during the winter and pasture renewal programmes should ensure soil infiltration rates are maintained.

NZDRI report soil physical properties of the irrigation farms are similar to other allophanic soils in the Waikato (NZDRI, 1999).

5.3 Soil chemical factors

As with many land treatment facilities, nutrient overloading is a problem on the LNZ irrigation farms. Nitrogen application and consequently nitrate leaching to groundwater is a major concern at many land treatment sites in New Zealand. At the LNZ site, nitrogen loading rates are approximately 475 kg ha^{-1} yr^{-1}. Average pasture uptake of nitrogen is approximately 575-710 kg ha^{-1} yr^{-1}, based on pasture dry matter production of 14 400 kg ha^{-1} yr^{-1} and a nitrogen pasture content of 4.0-4.9%. If the grazing animal was excluded from the current land treatment system, there is scope for a 'cut and carry' system with a good match between nitrogen inputs and harvest. It is important to note that heavy vehicles involved in harvesting will have an effect on soil structure and strength. As the grazing animals are part of the land treatment system, it is likely that there is leaching of nitrate to groundwater, under urine and dung patches. On heavily stocked pasture, animal returns of dung and urine can contribute anything up to 200 kg N ha^{-1} yr^{-1}. Since groundwater monitoring was first undertaken in 1991, there has been a general decline in nitrate groundwater concentrations in both the 'control' and 'impact' bores on the three irrigation farms. Concentrations in the 'impact' bores have decreased to levels similar to the 'control' bores, indicating the effluent irrigation is having little effect on nitrate groundwater concentrations. This would suggest that the current land treatment system is capable of assimilating the nitrogen contained in the effluent.

In terms of over-supply of nutrients, under the current conditions, phosphorus, potassium and sodium may pose greater economic and environmental concerns than nitrogen.

Phosphorus loading rates are approximately 435 kg ha^{-1} yr^{-1}. With a pasture phosphorus content of 0.4-0.5%, pasture uptake of phosphorus is in the vicinity of 60-70 kg yr^{-1}. Therefore, annually there is over 350 kg ha^{-1} of phosphorus to be adsorbed by the soil. This is reflected in the large Olsen P values (up to 350 μg cm^{-3}) and low P retention values (as low as 29%) at the 0-30 cm depth for paddocks irrigated for 40 years. Below 30 cm, effluent irrigation has had little effect on Olsen P and P retention values. This indicates that the soil has the ability to store more phosphorus. This is verified by the high P retention capacity as measured by the phosphate adsorption isotherms for the 40 year study paddocks, at the 30-45 and 45-60 cm soil depths.

When discussing phosphorus dynamics, the depth to the water table must be considered. Limited data from the Taranaki Regional Council suggest that the water table is, at least approximately 2.0 m below farms No.1 and No.2 and at least approximately 1.5 m below farm No. 3. This would indicate that there is still a very large capacity for phosphate fixation in these soils, before there is a threat of the water table coming in contact with soils saturated with phosphorus.

The breakthrough curve (Figure 4.7) indicates that when soil Olsen P levels reach 130 μg cm^{-3}, LNZ soils are saturated with P and no longer adsorb phosphorus strongly, leading to a dramatic increase in soil solution P levels. If the water table came into contact with soil saturated with P, the likelihood of P entering groundwater would be very high. This is why an Olsen P level of 130 μg cm^{-3} is considered as phosphate saturation for LNZ soils. The Olsen P values at 15-30 cm depth (Table 4.21) for paddocks irrigated with effluent for 30 years are 136 μg cm^{-3} and paddocks irrigated with effluent for 14 years at the 7.5-15 cm depth are 150 μg cm^{-3}. Therefore, as an approximate guide, it appears 1 cm of soil per year becomes saturated with phosphorus, to an Olsen P level around 130 μg cm^{-3}. In addition to this observation, an approximate estimate can be made to quantify the amount of phosphorus that can be further applied to LNZ irrigation farms soils, before the soil Olsen P levels reach 130 μg cm^{-3}.

Approximate calculations (Refer to Appendix 3) were made based on the limited information available. Calculations were based on the assumption that soils would not strongly adsorb phosphorus, at Olsen P values above 130 μg cm^{-3}.

The calculated figures are less than those interpolated from the Olsen P values. The difference in values can be attributed to the calculations not taking into account the amount of P adsorbed in the topsoil (soil with Olsen P values above130 $\mu g\ cm^{-3}$).

Table 5.1 Amount of phosphorus that can be applied to soils before phosphate saturation occurs at or near the depth to the water table

	Paddocks with previous years of effluent irrigation				
Soil depth	0	9	14	30	40
(cm)	(tonnes ha^{-1})	(tonnes ha^{-1})	(tonnes ha^{-1})	(tonnes ha^{-1})	(tonnes ha^{-1})
60	7.5 (17)	6.7 (15)	5.8 (13)	3.5 (8)	3.7 (9)
100	13.0 (30)	12.0 (28)	11.3 (26)	8.2 (19)	9.1 (21)
150	19.9 (46)	18.8 (43)	18.2 (42)	15.5 (36)	15.8 (36)
200	26.8 (62)	25.5 (59)	25.1 (58)	22.9 (53)	22.6 (52)

() The figures in brackets are the number of years before phosphate saturation occurs, based on a loading rate of 435 kg ha^{-1} yr^{-1}.

This indicates that with increasing soil depth, increasing amounts of P can be loaded before it reaches the P saturation level. Therefore, as long as the water table is kept below 200 cm depth it will take a very long period to cause any contamination of groundwater with P.

While collecting soil samples from paddock 32 on farm No.1 during September, the water table was within 40 cm of the soil surface in low-lying areas. Therefore, at certain times of the year, under paddocks on the western side of Dunnes Creek and potentially under any area with impeded drainage, the water table may come in contact with soil that is saturated with phosphorus and soil solution with high phosphorus concentrations. This would more than likely to result in phosphorus entering the groundwater system.

Phosphorus inputs are a key factor in the eutrophication of surface waters, because P is usually the most limiting nutrient for growth of phytoplankton which obtain their N requirements through biological N fixation. Only very small concentrations of dissolved P are required to cause accelerated growth of aquatic biota and cause problems. Up to 6 kg of P ha^{-1} yr^{-1} can be lost from dairy pasture runoff (Sharpley and Syers, 1979), which is 12-30 times higher than the natural rates of phosphorus loss which occur during the development of some soils. Due to the extremely high concentrations of phosphorus in the topsoil on LNZ irrigation farms, surface runoff of either recently irrigated effluent or ponded rainfall is likely to contain large quantities of P.

There are various factors that affect the amount of phosphorus lost from soil to surface and subsurface runoff. The overriding factor affecting P loss appears to be the amount of phosphorus in particular the amount of phosphorus in the top 2 cm (Blennerhassett, 1998). At the LNZ irrigation farms, Olsen P levels in the top 2 cm are likely to be very high. Soil moisture levels also have an influence on P loss. As the soil moisture decreases over spring and summer, there will be an increase in the P concentration in soil solution and thus an increase in the potential for P loss (Blennerhassett, 1998). Unfortunately, this coincides with the time of the year when eutrophication is a greater problem. During the summer months there is also high intensity rainfalls which are likely to cause large runoff events.

At the LNZ site, due to the extremely high concentrations of phosphorus in the irrigated soils, any runoff created is likely to contain high concentrations of P. An advantage of the LNZ paddocks is their relatively 'flat' slope. This will reduce the amount of runoff and the velocity of surface runoff. There are some measures that can be taken to minimise runoff volumes and surface runoff entering waterways. The vegetation cover has an influence on the amount of sediment that is suspended in runoff. An even cover of vegetation has the ability to decrease the velocity of the raindrops and therefore reduce the impact of raindrops on soil aggregates and the depth of the rainfall-soil interaction (Agassi *et al.*, 1985).

Minimising the amount of P-enriched surface runoff reaching surface waters is the most important technique to reduce surface and subsurface runoff on the LNZ irrigation farms. This is through the use of riparian strips. LNZ is well advanced in it's riparian planting programme. Continuation of this is necessary to ensure all waterways are surrounded with riparian zones.

The TRC has observed runoff from the irrigation farms, particularly on farm No.1 (TRC, 1988; TRC, 1994; TRC, 1995). However, since the change in management philosophy occurred in 1995, there have been few, if any reports of incidences of surface runoff to waterways. This, combined with the riparian planting regime, which has been progressing for the last three years, should ensure surface runoff to waterways is likely to have minimum impact on water quality.

Sodium loading rates are approximately 720 kg ha^{-1} yr^{-1}. With a sodium pasture content of 0.25-0.39%, pasture uptake of sodium is in the vicinity of 35-55 kg ha^{-1} yr^{-1}. High sodium concentrations are likely to cause deterioration in soil structure and soil strength, through deflocculation of clay particles. As sodium is a weakly held cation, it is also very susceptible to leaching. This is reflected in the elevated groundwater sodium concentrations, particularly under farms No.1 and No. 2. Groundwater concentrations of sodium on all three irrigation farms are higher in the 'impact' bores than the 'control' bores. Since monitoring of these bores began in October 1991 there is a trend of increasing sodium concentrations with time. TRC (1999) report the magnitude of the increase in sodium may be of concern in terms of the effects of saline water on pasture and soil quality. However, the saline effects continue to be mitigated by annual applications of lime, shallow subsoiling of paddocks and pasture renewal programmes. Lime application has the disadvantage of inducing the movement of Na^+ to groundwater.

Potassium loading rates are approximately 1570 kg ha^{-1} yr^{-1}. With a potassium pasture content of 3.4-4.2%, pasture uptake of potassium is in the vicinity of 490-605 kg yr^{-1}. Under the current dairying land use, leaching of potassium is exacerbated by the grazing animal. The effect of potassium has been shown to be significant with respect to nutrient related disorders in stock. The greatest leaching losses will occur during winter when the soils are at or near field capacity. Losses by leaching can account for up to 50% of the potassium added in dung and urine on pasture soils, and is approximately 100 kg ha^{-1} yr^{-1}. This is because the high concentration of potassium in each urine patch cannot be completely recovered by the plant. A 'cut and carry' system (exclusion of grazing animals) would help to overcome potassium leaching through no deposits of dung and urine and an increase in K uptake in silage and hay harvests.

Under any land treatment system, sodium and potassium will be applied in excess amounts. Therefore, it may be prudent to look at ways of decreasing the sodium and potassium concentrations of effluent before irrigation occurs.

With the addition of some nutrients in excess amounts and others in less abundance there is the possibility of creating a nutrient imbalance in the soil leading to nutrient related disorders in animals. LNZ have identified the necessity to add 1 tonne of lime ha^{-1} $year^{-1}$ to counter sodium and potassium effects on soil structure and mineral balances. The latest NZDRI report (NZRI, 1999) recommends lime or alternatively gypsum applications should continue at 1 t ha^{-1} (1.6 t gypsum ha^{-1}) on farms No.1 and No.3 and an increased application of 1.5 t ha^{-1} of lime or 2.4 t ha^{-1} gypsum should be used on farm No.2.

NZDRI (1999) report soil chemistry on the LNZ irrigation farms is typical of spray irrigation sites around New Zealand. NZDRI experience at other irrigation sites has shown that phosphorus accumulation has no adverse effects on farm production or stock health and that pH tends to rise to around 7.5. Although high levels of P induces deficiencies of other elements such as zinc, there is no evidence in the literature to suggest that high soil P status poses a threat to animal health.

5.4 Soil biological factors

From the limited soil biological measurements undertaken for this study it would appear effluent irrigation has had little detrimental or beneficial effect on soil biological properties, apart from an increase in individual earthworm biomass in irrigated paddocks.

5.5 Pasture performance

LNZ identified poor pasture production during November, which is sometimes accompanied by the yellowing of the vegetation, as a concern. One scenario to account for this phenomenon is that the soil dries out during this period and due to the relatively shallow rooting system, a nitrogen shortage causes the yellowing of the pasture. During the study, the yellowing of pasture was not observed. This may be due to the large rainfall recorded in November (291 mm) and January (165.5 mm) meaning the soil did not dry out as quickly. The use of nitrogen in this period may overcome this problem.

Pasture production is enhanced in irrigated paddocks. However, the pasture cover tends to become more sparse with time. This can have detrimental effects on soil structure, due to direct impact of irrigated effluent on soil aggregates and treading damage caused by grazing animals. The pasture renewal programme should be continued so as to mitigate this problem.

5.6 Wider environmental considerations

Wider environmental considerations should take into account any off-site impact the LNZ land treatment system may have. The most obvious factors are water quality and the recognition that farm produce becomes part of the human food chain. Groundwater quality monitoring at LNZ irrigation farms has been undertaken since the early 1990's. The latest TRC report (TRC, 1999) also discusses the possibility of some spatial synoptic surface water quality monitoring during the 1999-2000 period. At present the dairy farms operate as seasonal milk suppliers. Therefore, produce directly linked with the land treatment system is entering the human food chain. Obviously this has been happening for many years. However, it appears more stringent guidelines for land treatment of industrial effluent may prevent produce entering the food chain. Alternative land treatment systems would need to be considered at this point.

5.7 Alternative crops

Traditionally, land treatment systems on dairy farms have used pasture-based vegetation. There are a number of potential alternatives to growing pasture, including agricultural and forest crops. Various cash crops (cereals, seed crops and vegetables) and forage crops have been used for land treatment systems (Carnus *et al.*, 1994). However, few of these alternative crops can remove the same amount of nutrients as pasture and tolerate wet conditions. Many also require intensive management and often encounter periods in which effluent cannot be irrigated onto the paddock.

A further alternative land treatment option is to apply effluent to soil growing trees. Short-rotation forests (SRF) are being used for land treatment of various effluents in commercial and research sites around the world (Pertu, 1993; Myers *et al.*, 1998; Labrecque *et al.*, 1998; Guo and Sims, 1998). The short rotations take advantage of the ability of these species to grow back from their stumps (coppice) after harvest. Thus maintaining the fast, initial growth rates of these species. The land treatment potential stems from the application of water and nutrients, which may meet the demands of the crops' high growth rates (Nicholas, 1997).

SRF offer certain advantages over a pasture based land treatment system. Due to the greater evapotranspiration rate of trees in comparison to pasture, significantly lower drainage volumes are produced (Roygard, 1999). Also, due to the greater rooting depth of trees, nutrients and water can be taken up from greater depth, reducing leaching of nutrients. A further advantage of SRF is they are not part of the human food-chain.

The main disadvantage of SRF is that during the coppicing period, ideally no effluent should be applied in order to prevent leaching and surface runoff of nutrients. However, at the LNZ site it is not possible to withhold effluent application during the regrowth period. In addition, the current effluent irrigation system would need to be altered so effluent could be applied to trees.

Tree plantations of mixed species, combined with pasture may be a potentially valuable 'agro-forestry' approach to land treatment systems (Carnus *et al.*, 1998). This may be suitable on farm No. 1, where trees could be planted in the paddocks on the western side of Dunns creek, due to the wetness of these paddocks.

5.8 Management

Very little information pertaining to crop and animal management that is recorded in the literature has direct relevance to New Zealand because New Zealand soils, climate and pastoral farming systems are different from those of other countries. Most plants manage their irrigation sites based on local experience, gained by trial and error.

CHAPTER 6

Conclusion

6.1 Introduction

- Effluent composition showed extreme variability over short time intervals for all commonly measured effluent characteristics. LNZ effluent is characterised by high suspended solids, BOD_5, COD, total potassium, total P and total sodium and low pH.

6.2 Effect of effluent on soil physical properties

- The irrigation of effluent has altered some of the physical characteristics of the irrigated soil. The most appreciable changes have occurred at the 0-7.5 cm soil depth. There is a general trend of greater effects on soil physical properties, with increasing years of effluent irrigation.
- The bulk density of the topsoil in the irrigated paddocks is very low. There is a strong trend of decreasing soil bulk density with increasing years of effluent irrigation at 0-15 cm soil depth. Below 20 cm, bulk density values are very similar across all study years and sampling depths.
- The irrigation of effluent has had no consistent effect on the infiltration rate of the irrigation study paddocks. The 14 year study paddock had an infiltration rate of approximately 9 mm/hr, which was significantly lower ($P \leq 0.05$) than all other study paddocks. The 9 year study paddock had an infiltration rate of approximately 160 mm/hr, which was significantly greater ($P \leq 0.05$) than all other study paddocks. The 0, 30 and 40 year study paddocks had similar infiltration rates of 95, 80 and 85 mm/hr, respectively.
- Soil moisture contents in the 14, 30, and 40 study year irrigated paddocks were greater than the non-irrigated study paddocks throughout the year. There is a general trend of increasing soil moisture content with increasing years of effluent irrigation.

- Penetration resistance is significantly lower ($P \leq 0.0001$) on irrigated study paddocks compared to non-irrigated paddocks. There is a general trend of decreasing penetration resistance with increasing years of irrigation.
- All irrigated study paddocks had similar or slightly greater aggregate stability than the non-irrigated paddocks

6.3 Effect of effluent on soil chemical properties

- The irrigation of effluent has markedly increased the level of soil chemical fertility on the land treatment area. Increases of some nutrient levels have not only occurred in the 0-7.5 cm soil depth, but further down the soil profile. Generally, there is a trend of increasing nutrient concentrations with increasing years of effluent irrigation.
- Total carbon levels have increased slightly in all irrigated study paddocks. The largest increase in total carbon was measured at the 15-30 cm soil depth. This indicates that there was a movement of the soluble and fine particulate carbon added through effluent irrigation down the soil profile. The 14 and 40 year study paddocks have significantly greater ($P \leq 0.05$) total carbon in the soil profile than the non-irrigated study paddocks. Below 30 cm, effluent irrigation has had little effect on total carbon levels.
- Olsen P values have increased significantly ($P \leq 0.05$) in paddocks irrigated with effluent compared with non-irrigated paddocks at 0-7.5 cm depth. The largest increases in Olsen P were measured in the 30 and 40 year study paddocks at the 7.5-15 cm depth. To a depth of 30 cm, there is a general trend of increasing Olsen P with increasing years of irrigation. Irrigation has had little impact on Olsen P values of soil at a depth below 45 cm.
- Phosphate retention has decreased significantly ($P \leq 0.05$) on irrigated paddocks compared with non-irrigated paddocks at the 0-7.5 cm depth. To a depth of 30 cm, there is a general trend of decreasing P retention, with increasing years of irrigation.

- Effluent irrigation has increased the null point concentration of the irrigated paddocks (30 and 40 years) from approximately 1-2 ug/ml to approximately 100 ug/ml, indicating phosphate desorption is likely to occur in the 0-15 cm. There is a trend of increasing null point concentration with increasing period of effluent irrigation. Below, 30 cm effluent irrigation has had little effect on null point concentration, indicating these soils have the ability to adsorb additional phosphate.
- Above an Olsen P value of approximately 150 $\mu g/cm^3$, the null point concentration of irrigated soils is increased dramatically. This indicates that P desorption is likely to occur at around 150 $\mu g/cm^3$.
- Up to 130 μg Olsen-P cm^{-3} of soil, P was retained strongly in the soil and there was a negligible concentration of P in the soil solution. Above this P concentrations in soil solution were significantly higher, enhancing the potential for leaching losses.
- Total nitrogen levels have increased slightly in irrigated paddocks compared with non-irrigated paddocks, with the largest increases occurring in the 0-15 cm soil depth. Below 30 cm, effluent irrigation has had no significant effect on total N levels in irrigated study paddocks.
- In general, both nitrate and ammonia levels have increased on irrigated paddocks compared with non-irrigated paddocks. Both the 14 and 40 year study paddocks have large nitrate concentrations below 30 cm, indicating NO_3^- is moving down the soil profile. There was no significant difference in the NH_4-N at lower depths between the effluent irrigated and non-irrigated paddocks suggesting that ammonium is not moving down the profile.
- Exchangeable sodium and potassium levels have increased significantly ($P \leq 0.05$) on irrigated study paddocks compared with non-irrigated study paddocks at 0-7.5 cm depth. This may lead to soil physical deterioration. The greatest increases in sodium and potassium levels were measured at the 0-15 cm depth. Below 45 cm, effluent irrigation has had little effect on sodium and potassium levels.

6.3 Effect of effluent on soil biological properties

- Earthworm populations across the irrigated study paddocks are extremely variable. All irrigated paddocks have greater earthworm biomass/m^2 than non-irrigated paddocks. The mean individual earthworm mass is greater for paddocks receiving effluent, 0.62 g/earthworm compared to 0.34 g/earthworm for non-irrigated paddocks.

6.4 Effect of effluent on water quality

- The extensive use of all three farms for spray irrigation of effluent has impacted on shallow groundwater to varying degrees, raising sodium, nitrate, conductivity and occasionally filtered COD levels in the groundwater of most of the impact bores. However, the groundwater is not used for a potable supply and since the introduction of new management philosophies in 1995, groundwater nitrate levels have decreased.

6.5 Effect of effluent irrigation on pasture

- The irrigated study paddocks have similar values for most pasture quality parameters analysed. There is considerable difference between the dietary cation-anion difference (DCAD) of the irrigated and non-irrigated study paddocks. Animal diets with high DCAD values tend to increase the incidence of milk fever mainly due to Ca deficiency. This suggests that there is a need to supplement the Ca levels in the forage.
- Nutrient status of the pasture on the irrigation farms is in the normal to high range recommended by AgResearch.

6.6 Grazing trial

- Grazing of pasture directly after effluent irrigation had little effect on soil physical properties.

6.7 Bare Patches

- Bare patches on farm No. 2 have extremely low bulk densities (0.42 g cm^{-3}) and comparable soil chemical characteristics to paddocks irrigated for 40 years.

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Appendix 1

Calculations for mass balance of applied carbon

Solid content of effluent	= 1.77 % (Table 4.1)
C content of solid material	= 38.7%
Effluent loading rate	= 5 000 m^3 ha^{-1} yr^{-1}
C loading rate	= 5000 x 0.0177 x 0.387
	= 34.2495 m^3 ha^{-1} yr^{-1}
	= 34 250 kg ha^{-1} yr^{-1}
Over 9 years	= 308 250 kg ha^{-1}

Assumption: all C is retained in the 0-60 cm depth

Soil volume for 0-60 cm depth	= 10 000 m^2 x 0.60 m
	= 6000 m^3
C addition	= 308 250 kg ha^{-1}
	= 308 250 kg ha^{-1} / 6 000 m^3
Calculated C addition	= 51.375 kg m^{-3}
C measured in the soil	= 18.60 kg m^{-3} (approximate mean for control soils)
	= 22.13 kg m^{-3} (approximate mean for 9 years)
C increase from effluent	= 22.13 – 18.60
	= 3.53 kg m^{-3}
Percentage of C accumulated	= 3.53 / 51.375
	= 6.9%

Appendix 2

Calculations for mass balance of applied phosphorus

Annual P loading	= 435 kg ha^{-1}
Over 9 years	= 3915 kg ha^{-1}

Assumption: all P is retained in the top 0-15 cm

Soil volume for 0-15 cm depth	= 10 000 m^2 x 0.15 m
	= 1500 m^3
	= 1500 x 10^6 cm^3
P addition	= 3915 kg ha^{-1}
	= 3915 kg ha^{-1} / 1500 x 10^6 cm^3
	= 3915 mg/1500 cm^3
Calculated P addition	= 2.61 mg cm^{-3}
P measured in the soil	= 2.86 mg cm^{-3} (approximate mean for control soils)
	= 4.215 mg cm^{-3} (approximate mean for 9 years)
P increase from effluent	= 4.215 – 2.86
	= 1.355 mg cm^{-3}
Percentage of P accumulated	= 1.355 / 2.61
	= 52%

Appendix 3

Calculations for phosphorus loading before P saturation occurs

Olsen P saturation value (calculated from Figure 4.7) = 130 $\mu g\ cm^{-3}$

Measured Olsen P for 0-7.5 cm for control soil (Table 4.21) = 73 $\mu g\ cm^{-3}$

Olsen P required to reach the saturation level ($\mu g\ cm^{-3}$) = 130 – 73

= 57 $\mu g\ cm^{-3}$

Olsen P required to reach the saturation point ($kg\ ha^{-1}$)

Volume of soil = 10 000 m^3 x 0.075 m

= 750 m^3

Olsen P = (750 x 57) / 1000

= 43 $kg\ ha^{-1}$

Total P required to reach saturation point ($kg\ ha^{-1}$)

Percentage contribution of effluent P to Olsen P

$$= \frac{\text{change in Olsen P}}{\text{change in total P}} \times 100$$

$$= \frac{251 - 73\ (\mu g\ cm^{-3})}{5.40 - 3.35\ (mg\ cm^{-3})}$$

$$= \frac{0.251 - 0.073\ (mg\ cm^{-3})}{5.40 - 3.35\ (mg\ cm^{-3})}$$

= 8.7%

Amount of P required is = 43 $kg\ ha^{-1}$ / 0.087

= 494 $kg\ ha^{-1}$