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THE EFFECT OF LAND DISPOSAL OF DAIRY FACTORY WASTES ON SOIL PROPERTIES

A thesis presented in partial fulfilment of
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ABSTRACT

Many New Zealand dairy factories dispose of their wastewater by spray irrigating onto pasture. Little is known, however, about the effects of this disposal on soil properties. Research was undertaken at three pasture disposal sites in order to determine whether certain soil property changes may have occurred as a result of the wastewater treatment. Of particular interest were those properties related to water movement.

Laboratory studies using 'undisturbed' soil cores indicated that dairy factory wastewater can impede soil water movement. A single application of simulated whey effluent resulted in approximately a 50% decrease in saturated hydraulic conductivity (K) within two days. This reduction was observed to be caused by a combination of both physical and biological blockage processes. With repetitive doses of effluent a K decrease of over 99% was induced in some cores. Several cores, particularly those containing earthworms, showed signs of recovery, and in some cores the final hydraulic conductivity value was greater than the initial value.

Analyses of soil samples from the disposal and control sites at Te Rehunga and Tokomaru suggest that fifteen years of wastewater irrigation have resulted in marked changes in soil physical, chemical and biological properties. Total carbon and nitrogen levels were found to be significantly higher at the disposal site; for the Te Rehunga site, the differences in the organic matter level down to 600mm represented an increase of 250 000 kg ha^{-1}.

Water balances for the Te Rehunga and Longburn sites indicate that, in the absence of wastewater, pasture is likely to be water stressed on average for approximately forty days per year. The water balance also shows that deep percolation will be greatly increased by the wastewater application. The period of maximum deep percolation loss is likely to be September to October at both the Te Rehunga and Longburn disposal sites.

The major site management problems encountered at the disposal sites examined occurred as a result of poor soil drainage, pasture burning and pasture pulling. An infiltration problem was observed at the Longburn site and the recently established disposal site at Tokomaru, with two major causes of the low infiltration rate appearing to be blockage from the effluent and pugging; these observations illustrate the need for controlling the effluent application rate, the suspended solids level in the wastewater, and the stock grazing pattern, in order to minimise site drainage problems. A drainage problem over the winter-spring period at Te Rehunga was due to a high groundwater table. Pasture burning was observed at all three disposal sites. The pasture pulling problem at Te Rehunga is the only cited example of such a problem occurring at a dairy factory disposal site.

Observations made at the established Te Rehunga and Tokomaru disposal sites show that long term spray irrigation of dairy factory wastewater can occur without inducing undesirable soil property changes. It appears as though considerable benefit can be gained from the wastewater irrigation, particularly in reducing the incidence of water stress in the pasture and decreasing the requirement for fertilizer.

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CHAPTER ONE

INTRODUCTION

CHAPTER ONE

INTRODUCTION

The dairy industry is vital to the economy of New Zealand. The industry earned for New Zealand \$2 000 million over the six-year period 1971-1977, averaging about 19% of this country's export earnings. In the 1976-1977 season, 700 000 tonnes of dairy produce were exported, earning more than \$400 million.

Associated with the large quantities of marketable goods from the dairy industry is an immense volume of waste material requiring disposal. Little information is available on the actual quantity and composition of the effluent from dairy factories. However, a survey conducted by Marshall (1975) revealed that the volume of effluent produced by New Zealand dairy factories varied from a half to three times the milk intake volume. Since it has been noted that peak daily milk intake at some factories can exceed 1200 m^3 per day (Parkin and Marshall, 1976), it is evident that large volumes of wastewater are required to be disposed of daily.

The composition of dairy factory effluent is related to the factory product and the degree of technology employed but in general dilutions of milk, milk products and by-products, cleaning agents and wash-down waters are the major constituents of the wastewater. Fluctuations in time, both throughout the day and over a season, make it difficult to obtain representative samples for composition analysis (Pound and Crites, 1973). Of the materials discharged by dairy factories whey is generally regarded as the most potentially harmful to the environment (United States Environmental Protection Agency, 1971). Whey is a by-product from cheese and casein manufacture, both of which New Zealand produces in large quantities. Most of the cheese whey produced is dried for lactose manufacture, or is used as pig feed. As yet, however, little use is made of casein whey.

In the past effluent disposal was of no great concern to the New Zealand dairy industry. Factories were in most cases sited near to a watercourse which served as a sink for the produced wastewater. Two factors have resulted in the present need for alternative disposal methods. Firstly, the volume of effluent discharged into watercourses (both from dairy factories and other sources) has increased, thus placing extra strain on the assimilation capacity of the receiving water. The increase in the volume of waste per factory is indicated by the change in product output per factory. Average output from individual cheese and butter factories, for example, has more than doubled since 1960 (New Zealand Official Yearbook, 1977). Secondly, there has been an increase in public awareness of water quality degradation which has resulted largely from increased public utilisation of waterways. This change in public concern has led to the introduction of legislation designed to combat environmental degradation; the first really effective legislation was the New Zealand Water and Soil Conservation Act of 1967.

As with other waste dischargers the dairy industry is finding that effluent discharge to waterways is no longer permissible and that alternative disposal methods must be employed. Parkin and Marshall (1976) reported that a number of dairy factories have adopted alternative disposal methods; they noted that the number of factories discharging their wastes to natural waterways had been reduced by 50% over the six year period 1970-1976.

The dairy factory faced with the problem of seeking an alternative to waterway discharge may be confronted by a number of choices. The treatment method selected would generally be the alternative most economically viable, providing it results in an environmentally acceptable level of waste treatment. In certain cases the most viable treatment method could be a utilisation process, whereby the effluent is able to be processed into a marketable product. Lactose manufacture from cheese whey is one example of a utilisation process. In other instances, however, the most suitable means of waste treatment could be a disposal process whereby the environmentally harmful components in the waste are removed or transformed to a less harmful state. Biological and chemical oxidation, trickling filters, activated sludge, oxidation ponds and spray irrigation methods are the major disposal techniques used around the world. Of these techniques, biological and chemical disposal methods are of particular importance overseas (United States Environmental Protection Agency, 1971). On the New Zealand scene, however, few dairy factories have employed these chemical or biological disposal methods, with spray irrigation appearing to be a more feasible alternative.

The popularity of spray disposal methods for dairy factory wastes in New Zealand appears to be a result of several factors. Firstly, New Zealand dairy farming is strongly seasonal (McDowall and Thomas, 1961), and since the size of a treatment plant is generally based on the peak effluent loading, plant efficiency will be lowered. A low efficiency in such treatment plants will compound the already high costs involved. Secondly, most New Zealand dairy factories are sited in rural areas, thus there is generally no great problem involved in obtaining land for the irrigation system. Land cost and availability in many overseas countries is frequently the major limitation in development of spray disposal systems (Pound and Crites, 1973). Thirdly, the efficiency of a biological or chemical treatment plant is markedly reduced by the presence of any whey or casein washwater in the effluent (United States Environmental Protection Agency, 1971); New Zealand produces a relatively large amount of casein, with over 70 000 tonnes of casein produced in the 1976-77 season (New Zealand Official Yearbook, 1977). Finally, the climate in most New Zealand regions is well suited for spray disposal systems. Whereas waste irrigation may be carried out all year round in New Zealand, this is not possible in many countries where frozen soil conditions are experienced. The seasonal rainfall and evapotransporation pattern in New Zealand coincides reasonably well with the seasonal pattern observed for dairy production, although hydraulic loading problems may be encountered over the September-October period when the soil's assimilative capacity is low and dairy production is at a peak.

Spray irrigation of wastewater is not unique to the New Zealand scene. Many countries, including Australia, Canada, Germany, France, India and the United States, have employed spray irrigation to dispose of a number of different types of liquid wastes. One of the first reports on the usage of spray disposal was in 1947, by the Hanover Canning Company of the United States (Pound and Crites, 1973). Since then the practice has shown rapid expansion and by 1970 over 2400 land waste disposal systems were in use throughout the United States alone (Evans, 1970).

The most common approach to the design and management of many former waste disposal schemes was to employ the principles developed for conventional irrigation practices, with little or no concern given to the solids component of the effluent. A change in concept has evolved, however, and today waste application to land is not merely considered from the disposal viewpoint but may also be viewed as a utilisation process whereby water and nutrients are supplied to the crop as a means of increasing production. It is important in the design and management of present day spray disposal systems to understand the effects that effluent application may have on soil properties, particularly those properties related to site management efficiency. To obtain information on soil property changes due to effluent application it is necessary to examine established disposal sites. For most sites the level of soil parameters prior to irrigation would be unknown. Hence it is necessary to deduce any changes which have occurred by using an untreated site, of the same soil type, for comparison.

Reports on the operation of spray disposal sites suggest that the hydraulic loading onto the site is the most important factor to consider in design (Canham, 1955; Pound and Crites, 1973). As a result any changes in the drainage characteristics of the disposal site are likely to have particular significance to efficiency of the spray disposal system operation. Consideration must also be given to the fate of the solid component of the wastewater, particularly the nitrogen fraction, since much recent interest has been shown in possible groundwater contamination following waste application to soils (Larson and Gilley, 1976; Adriano et al., 1975).

Spray irrigation is widely practised throughout New Zealand, hence research into the process, and the problems which may arise from it, is warranted. This thesis discusses the effects of the waste discharge on soil properties (particularly those properties pertaining to drainage), as well as the long term ability of the disposal site soil to accept effluent without incurring site management problems; since most spray disposal sites in New Zealand are also pasture - producing areas (generally dairying), site management considerations should relate to the farming situation as well as the disposal system operation.

CHAPTER TWO

SITE DESCRIPTIONS

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SITE DESCRIPTIONS

Three dairy factory spray disposal sites, the Tui Dairy Cooperative at Te Rehunga, the Manawatu Dairy Cooperative at Longburn and the Manawatu Dairy Cooperative at Tokomaru were examined. In addition to these three sites, samples for 'undisturbed' core studies were also taken from a Tokomaru silt loam and a Manawatu silt loam.

2.1 The Te Rehunga Disposal Site

This disposal site had received applications of dairy factory wastewater since 1963. The soil at the site was a Kopua silt loam, formed from alluvial and colluvial deposits of loess and greywacke. A description of this soil profile is given in Table A1.1.

2.2 The Longburn Disposal Site

The Longburn dairy factory consists of two separate units, one a milk-powder plant and the other a casein plant. Since most of the attention was focused on the casein plant effluent, only the disposal site receiving casein wastewater was examined. This site had been brought into operation for the first time in the 1977-78 season.

The soil at the casein waste disposal site was a Karapoti brown sandy loam, which is classed as a weakly leached non-accumulating river flat soil. The profile description of this soil is presented in Table A1.2.

2.3 The Tokomaru Disposal Site.

Most of the research at Tokomaru was conducted on an established disposal site which had received casein wastewater since about 1963. The soil at this disposal site was a Kairanga fine sandy loam which is classed as a slowly accumulating gley recent soil. The profile description of this soil is given in Table A1.3.

In addition to this established site the Tokomaru dairy factory has recently commenced spraying onto a new area, found to be a Kairanga peaty silt loam. This site was not as extensively studied as the previously mentioned sites, hence does not warrant a detailed profile description.

2.4 The Tokomaru Silt Loam

This soil is classed as a weakly leached, moderately to strongly gleyed, yellow grey earth. A detailed description of this soil profile is given by Pollok (1975).

2.5 The Manawatu Silt Loam

The Manawatu silt loam is termed a slowly-accumulating recent soil formed from alluvial material. A description of this soil profile is presented in Table A1.4.

CHAPTER THREE

METHODS OF SOIL ANALYSIS

CHAPTER THREE

METHODS OF SOIL ANALYSIS

3.1 Determination of Soil Physical Properties

3.1.1 Bulk density

Cylindrical metal samplers, 50mm in diameter and usually 50mm in length, were pressed into the soil above the desired sampling depth. The cylinders, containing a known volume of sample, were then withdrawn from the soil and material outside of the sample volume removed. The samples were oven-dried at 105C and weighed. Bulk density was calculated as the oven-dry mass of soil divided by the sample volume.

3.1.2 Particle density

The particle density, or density of the solid fraction of the soil, was determined by using a 50ml pycnometer. The general procedure described by Blake (1965) was employed, except for the use of reduced pressure in a vacuum dessicator to remove trapped air, instead of boiling.

3.1.3 Water retention properties of the soil

The pressure plate apparatus was used to determine the water retentivity (gravimetric water content) of soils at specific matric potential values. A general outline of the procedure is given by Richards (1965).

Three potentials, -0.1, -1.0 and -15 bar, were used to approximate "field capacity", the lower limit of readily available water and the "permanent wilting point", respectively. Since the structural properties of the soil have a marked effect on the water retention characteristics,"undisturbed" cores or aggregates were used at -0.1 and -1.0 bar potentials. At -15 bar potential the effect of structure is not so significant, and small aggregates packed to a height of approximately 10mm on the plate (each sample having an oven-dry weight of approximately 15g) were used. This restriction in sample height was imposed to ensure that hydraulic equilibrium was attained within the seven-day interval used.

From the gravimetric water content (w) values, it was possible to calculate the volumetric water content (θ) using

$$\theta = W \rho_b/\rho_w \qquad \ldots\ldots (3.1)$$

where ρ_b is the bulk density and ρ_w is the density of water. The "available water holding capacity" (AWC) was then computed as

$$AWC = \int_o^{z_r} (\theta_{FC} - \theta_{PWP}) \, dz \qquad \ldots\ldots (3.2)$$

where z_r is the effective rooting depth,

z is the depth of soil,

θ_{FC} is the volumetric water content at "field capacity"

and θ_{PWP} is the volumetric water content at "permanent wilting point".

3.1.4 Laboratory measurement of hydraulic conductivity

Unsaturated flow is a more common flow process in soils than is saturated flow. The measurement of unsaturated flow is extremely difficult however (Bertrand, 1965), and it is primarily for this reason that saturated hydraulic conductivity (K) was the measured parameter.

The procedure employed for the laboratory determination of K is described as follows: "Undisturbed" soil samples were obtained using a double cylinder sampler, as described by Lutz et al. (1947). Cores were collected when the soil moisture content was approximately at "field capacity" so that cracking or compaction effects would be minimised.

Once collected and transported to the laboratory the "undisturbed" cores were removed from the metal liner and coated with paraffin wax. To prevent collapse of the soil at the base a 75mm diameter filter funnel and gauze were waxed onto each core. Tissue paper was placed between the core base and the gauze. The completed apparatus is shown in Fig. 3.1.

Measurement of K in the laboratory was achieved using a "constant head" method which, as the name implies, involves maintaining a constant fluid

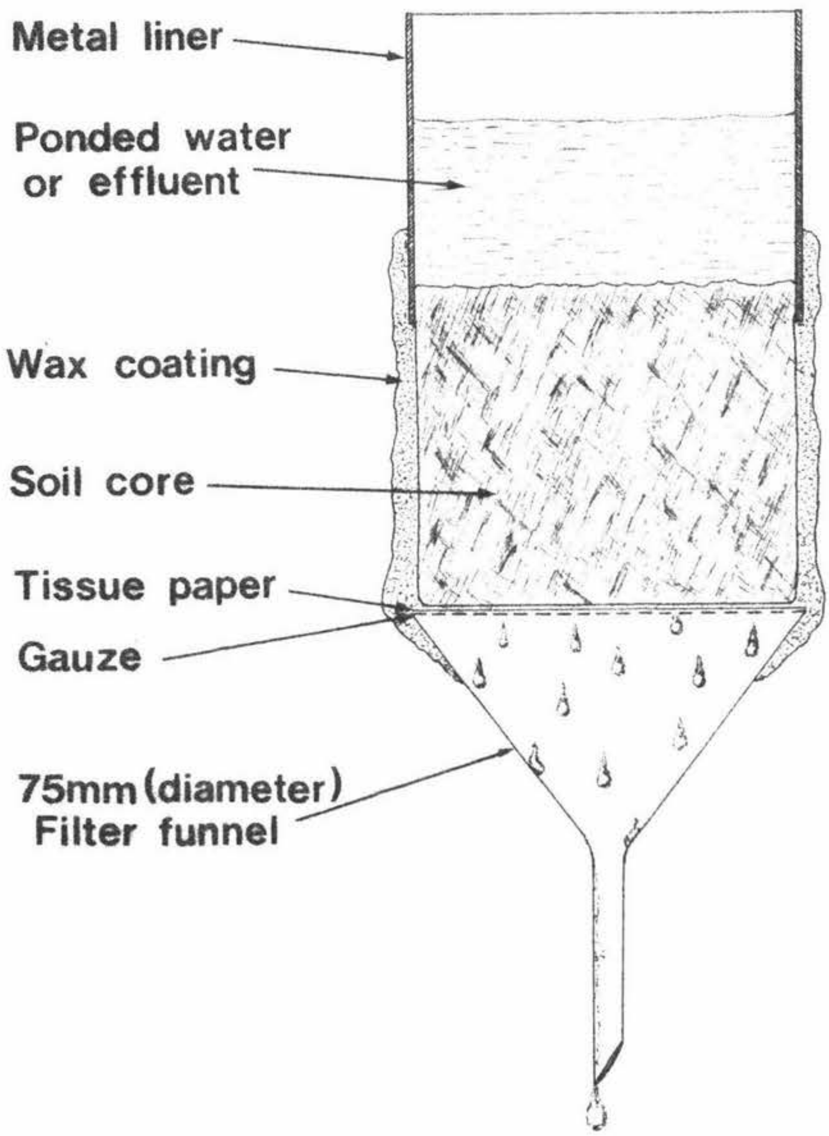

Fig 3.1 The apparatus used to study saturated hydraulic conductuvity in the laboratory

level in the metal liner over the measurement period. K is computed as,

$$K = (Q/A)\left[L/(L+H)\right] \qquad .....(3.4)$$

where Q is the flow rate (m^3 sec^{-1}),

A is the area of the core surface (m^2),

L is the length of the core (m),

and H is the height of the ponded fluid (m).

3.1.5 Field measurement of infiltration

A double-ring infiltrometer, as described by Bertrand (1965), was used for field infiltration measurements. The diameters of the inner and outer concentric rings used were 150mm and 380mm, respectively. The use of an outer ring was merely to provide a buffer effect, whilst the actual measurement was made by recording the fall of the water level in the central cylinder. Readings were taken at 1, 5, 10, 15, 30, 45, 60 and 90 minutes after the addition of water.

3.1.6 Dye tracing of soil macropores

Two staining solutions, one 5% (by weight) Rhodamine-B and the other 1% Methylene Blue, were employed. The dye was applied to the "undisturbed" soil cores (described in Section 3.1.4) following application of water, thus ensuring that the soil moisture content of the core was near saturation at the time of application.

3.2 Determination of Soil Chemical Properties

3.2.1 Organic carbon

The organic carbon levels were determined by wet oxidation with dichromate using the "Walkley Black" method (Allison, 1965). The mass of soil sample used was largely determined from the depth in the profile from which the sample was taken. As a rule, the sample weight varied from 0.3 grams in the surface layers to 2 grams at a depth of 600mm.

From the organic carbon percentage it was possible to estimate the organic matter percentage by using a conversion factor which assumes the average carbon content of organic matter to be 57% (Allison, 1965).

3.2.2 Total nitrogen

The total nitrogen content of a sample was determined using a semi-micro Kjeldahl method as described by Bremner (1965). Approximately 2 grams of finely ground air-dry sample was used. Digestion of the sample, to convert all nitrogen to the ammonium form, was achieved using a micro-Kjeldahl digestion block. A Hoskin's steam distillation unit was employed to determine the ammonium concentration in the digest.

3.2.3 Ammonium and nitrate

The analysis of ammonium-nitrogen (NH_4-N) and nitrate-nitrogen (NO_3-N) in soil extracts was achieved using a steam distillation method as described by Bremner and Keeney (1964). The soil extract for the inorganic nitrogen determinations was obtained by shaking 5 grams of soil sample and 50 ml of 2M KCl for a period of one hour. If distillation of the sample was to be delayed following KCl extraction, the soil extracts were filtered and stored at approximately 4C.

The NH_4-N and NO_3-N analyses were undertaken as soon as possible after sample collection so that any transfer of nitrogen between the various pools could be minimised.

3.2.4 Mineralisable nitrogen

Soil samples were air dried and then finely ground. Initial levels for NH_4-N and NO_3-N were determined as described in Section 3.2.3. Ten grams of the powdered sample, together with 30 grams of washed sand, was added to a 200ml plastic shaker bottle containing 6ml of water. Following shaking, the mixture was incubated at a temperature of 25C for fourteen days.

After incubation 100ml of 2M KCl was added to the soil-sand mix.

The samples were then shaken for one hour and filtered. NH_4-N and NO_3-N levels were determined as previously described.

To calculate the net mineralisable nitrogen, the pre-incubation NH_4-N and NO_3-N levels were subtracted from the sum of the post-incubation NH_4-N and NO_3-N.

3.2.5 Plant-available phosphorus

The test adopted as an index of plant available phosphorus is termed an 'Olsen's extractable phosphate' test and is described in detail by Olsen et al. (1954). In brief, this test involves shaking a known quantity of soil with a 0.5M sodium bicarbonate solution at pH 8.5 which is able to remove part of the phosphate on the soil solid phase. The level of phosphate which is brought into solution is then determined colorimetrically.

3.2.6 Soil pH

The pH of a soil sample was determined using a 1:2.5 soil:water slurry and a standard glass electrode pH meter as in the procedure described by Peech (1965). The pH measurement was made within twenty-four hours of sample collection.

3.3 Determination of Soil Biological and Microbiological Properties

3.3.1 Soil microbial respiration

The microbial activity in a soil sample was estimated using oxygen consumption rates obtained by soil respiration studies. For this determination, six respirometer units (as illustrated in Fig. 3.2) were constructed. The respiration vessels were adapted from 90mm diameter preserving jars having a capacity of approximately 0.5 litres. A disposable syringe (used for returning the manometer fluid level to equilibrium) and a length of glass tubing were fixed to each of the vessel lids. It was essential to ensure that the respiration unit was completely air tight. The respiration vessels were set in a water bath maintained at 25.5C $\pm$ 0.3C. A constant temperature was necessary for accurate results as the

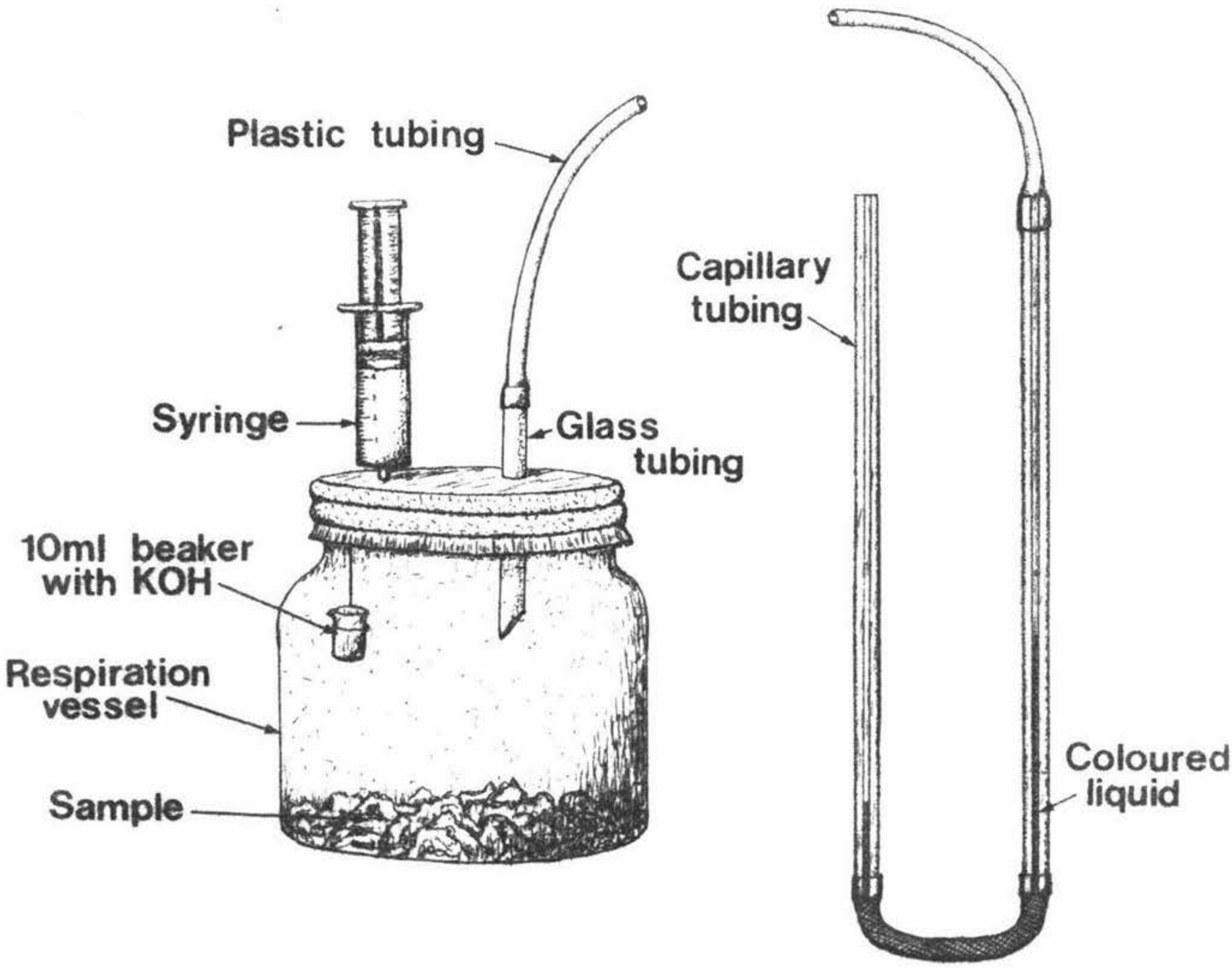

Fig 3.2 The Respirometry Apparatus

respirometer behaves as a thermobarometer. The manometer was made from 0.7mm diameter capillary tubing with a coloured solution as the indicating fluid. To remove any carbon dioxide produced during soil respiration, a 10ml beaker containing a 30% potassium hydroxide solution was attached to the inside of the respiration vessel. To increase the surface area of the potassium hydroxide solution a wick made from fluted filter paper was immersed in the beaker.

The soil sample was added to the respiration vessel either as large aggregates or as 2mm sieved soil. Following placement in the respirometer, the soil was equilibrated at the incubation temperature for 45 minutes. The rate of oxygen consumption was determined by measuring the rate of height change of the manometer fluid. Values for oxygen consumption, in moles of oxygen, was computed from the measured fluid height changes as

$$\Delta n = \left[P_1V_1 - (P_1 \rho_w gh)(V_1 - \pi r^2 h)\right] / (RT) \qquad \ldots\ldots (3.5)$$

where Δn is the number of moles of oxygen consumed in the specified time interval,

R is a gas constant ($8.3 J\ kg^{-1}\ mol^{-1}$),

T is temperature (K),

P_1 is atmosphere pressure (10^5 Pa),

ρ_w is the density of water ($10^3\ kg\ m^{-3}$),

h is the change in height of the manometer fluid (m),

r is the radius of the capillary tubing (m),

g is acceleration due to gravity ($9.8 m s^{-2}$),

and V_1 is the volume of the respirometry vessel plus plastic tubing and capillary tubing (m^3);

the derivation of this equation is shown in Appendix III.

3.3.2 Earthworm numbers

The worm count procedure involved collecting undisturbed soil cores and recording the number of earthworms found in each core after crumbling. The apparatus used for collecting the cores was the double cylinder sampler used for obtaining undisturbed cores (Section 3.1.4). Cores collected were 75mm in diameter by 110mm in length, giving a sample volume of 4.86×10^{-7} m^3.

Worm counting was carried out during May and June when it was considered that the majority of earthworms would be in the top 100mm of soil (Yeates, 1976).

3.4 Determination of Effluent Properties

3.4.1 Total and suspended solids

The percentage of total solids in the effluent solution was determined by evaporating a known mass of the solution to dryness at 105C and recording the final dry weight of remaining material. Approximately 0.1 to 0.2 kilograms of sample solution was used.

For the determination of suspended solids, the effluent sample was first filtered using Whatman Filter paper number one and then a known weight of the filtrate evaporated to dryness. The level of suspended solids, expressed on a percentage basis, was computed as the total solids less the dissolved solids.

3.4.2 Electrical conductivity

The electrical conductivity of an effluent sample was measured with a Radiometer conductivity meter (type CDM 3).

3.4.3 Carbohydrate

The total carbohydrate level in effluent samples and in soil leachate samples was determined using the Anthrone test procedure described by Loewus (1952). Carbohydrate levels in samples were determined colorimetrically by reading the absorbance at 600nm. The reading obtained for a sample was related to the standard curve values obtained for glucose; the carbohydrate level in the sample is therefore expressed on a glucose equivalent basis. In order to register on the standard curve (with a maximum value of 100 μg of glucose per ml of solution) it was found necessary to dilute the effluent and leachate samples by a factor of 50.

3.4.4 Simulation of whey effluent

Undiluted acid casein whey was collected directly from a dairy factory and diluted 10:1. This dilution gave a total solids value of approximately 0.6% and a total carbohydrate content of approximately 5kg m^{-3} of solution. Obtaining a representative sample for dairy factory effluent is difficult due to the wide variation in composition both throughout the day and over a season (United States Environmental Protection Agency, 1971). However, it would appear from the values cited for effluent composition (Pound and Crites, 1973), that this 10:1 dilution may provide a sample suitably representative for the purposes of this study.

CHAPTER FOUR

SOIL WATER BALANCE STUDIES

CHAPTER FOUR

SOIL WATER BALANCE STUDIES

4.1 Introduction

An excess soil water condition can markedly reduce plant growth and complicate site management (Luthin, 1966; Hudson et al., 1962). In order for a disposal site to operate efficiently, it is evident that the hydraulic loading onto the site must be considered in design and operation of the disposal system. Many workers consider that for a liquid waste disposal site the hydraulic loading is the principal design factor (Pound and Crites, 1973; Lawton et al., 1959; Gilde et al., 1971).

Water, like any other matter, must comply with the law of conservation of matter whereby it can neither be created nor destroyed. Because of this fact, a water balance can be derived for a soil in which the water input to a soil is balanced by the output plus any change in water stored in the profile. A basic soil water balance can be constructed for the plant rooting zone of the soil profile with the inputs being rainfall (M) and effluent irrigation (IR), and the outputs being evapotranspiration (ET) and runoff (RO). Equation 4.1 summarises this basic soil water balance where Δs is the change in water stored in the soil profile and RO includes both surface runoff and deep percolation.

$$M + IR = ET + RO + \Delta s \qquad ..... (4.1)$$

Soil water balances are extensively used in soil water studies. The area of interest relevant to this study, however, is their employment in spray irrigation scheduling. In water irrigation systems, water balances have been used to determine both the return interval for the irrigation and the application rate per dose (Jensen, 1973). The procedure used for normal irrigation may also be adopted for wastewater irrigation. The differing aims of wastewater irrigation, where disposal is generally considered to be the major objective, must however, be taken into consideration.

A water balance may be used to calculate the maximum return interval for the wastewater irrigation in order to prevent water stress conditions from occurring. The water balance could be used for site planning purposes primarily in determining the area required for the disposal site. This site planning will generally be based upon the period of greatest hydraulic loading, hence the water balance may also be of use in determining when this period occurs. However, since the quantity of wastewater irrigated in any one dose will be determined primarily by the quantity of liquid to be disposed of (as opposed to a water irrigation system in which the application rate is based upon plant requirements), water balance information may be of little use in the day to day operation of the system.

4.2 Procedure For The Construction of the Basic Soil Water Balance

The following description is of a daily water balance, whereby the nett daily input to the soil (M + IR = ET) is calculated. However, the procedure can just as easily be applied over a different time scale.

To construct the water balance it is necessary to know the initial water content. For this reason the starting point is usually taken after a heavy rainfall event in the winter-spring period when it is assumed that the soil profile is at field capacity. Field capacity is defined as the upper limit of available water storage in the root zone, occurring after free drainage has ceased. From this starting point the nett daily hydraulic input or loss from the soil profile is then added to, or subtracted from, the previous day's soil water storage value. When the net input results in a soil moisture content exceeding the field capacity value, the excess is considered to be lost from the profile as runoff; it is assumed in the water balance that the infiltration capacity of the soil is high enough to prevent any surface ponding from occurring (hence there is no surface runoff), so the runoff term in the water balance describes deep percolation losses.

Following a period of high evapotranspiration the total plant available water supply in the root zone may become effectively exhausted; the water content at which this occurs is termed the "permanent wilting point". Since evapotranspiration cannot reduce the soil water content beyond the permanent wilting point, any unsatisfied evaporative demand after the root zone has dried out to this point is referred to as a deficit. A sample water balance calculation is shown in Table A2.4. With spray irrigation, however, it is unlikely that a deficit will develop because of the constant irrigation input .

In order to construct the basic soil water balance it is necessary to obtain quantitative data for several parameters, namely: rainfall, wastewater application, evapotranspiration, soil water retentivity characteristics and crop rooting depth. Local meteorological data is commonly used for the rainfall and evapotranspiration determinations, although rain gauges and evaporation pans situated at the site may also be used if available. Effluent application is estimated either from factory production data coupled with the area of the disposal site, or by rain gauge measurement of the amount of wastewater applied to the soil.

The "waterholding capacity", or the water held between "field capacity" and "permanent wilting point" in the root zone, is often estimated from soil water retentivity characteristics. This estimation involves measuring the soil water content of specific matric potential values using the pressure plate apparatus described in Section 3.1.3. Soil bulk density values over the rooting depth are required in order to express the soil water content on a volumetric basis; the procedure used for bulk density determination is described in Section 3.1.1.

It is necessary to estimate the crop rooting depth so that the effective soil volume involved in water cycling can be determined. Rooting depth is often estimated by visual observation of the soil profile.

4.3 Results and Discussion

A daily water balance was constructed for the Te Rehunga site using rainfall data for the ten year period January 1967 to December 1977. A second balance for Palmerston North was supplied by the New Zealand Meteorological Service; this balance was derived from rainfall data over the thirty-four year period 1941-1975. For the purposes of this study, it was considered acceptable to use the Palmerston North water balance data at the Longburn and Tokomaru disposal sites; as explained in Section A2.1. Evapotranspiration and basic soil data necessary for the water balance determinations are presented in Tables A2.2 and A2.3, respectively.

The mean monthly runoff values for both computed water balances are shown in Figs. 4.1 and 4.2. In addition to the runoff occurring due to rainfall only, an estimation has also been made of the additional runoff resulting when effluent is applied. From the results it is seen that the maximum runoff in the absence of irrigation occurs in the May-October period. Following addition of the wastewater component, however, it is noted that runoff is greatest during the September-October period. It appears, therefore, that these two months are the critical design period for spray disposal systems. Computing the runoff values in Figs. 4.1 and 4.2 on a per annum basis it is found that the wastewater irrigation has doubled total annual runoff (from 800-1600 mm year^{-1}) for Te Rehunga, and trebled the runoff (from 280-860mm year^{-1})for the Longburn site.

Figs. 4.3 and 4.4 show the number of days per month on which the root zone is estimated to be at permanent wilting point (-15 bar) in the absence of irrigation; values for the lower limit of readily available water (-1 bar) have also been incorporated in the Te Rehunga water balance.

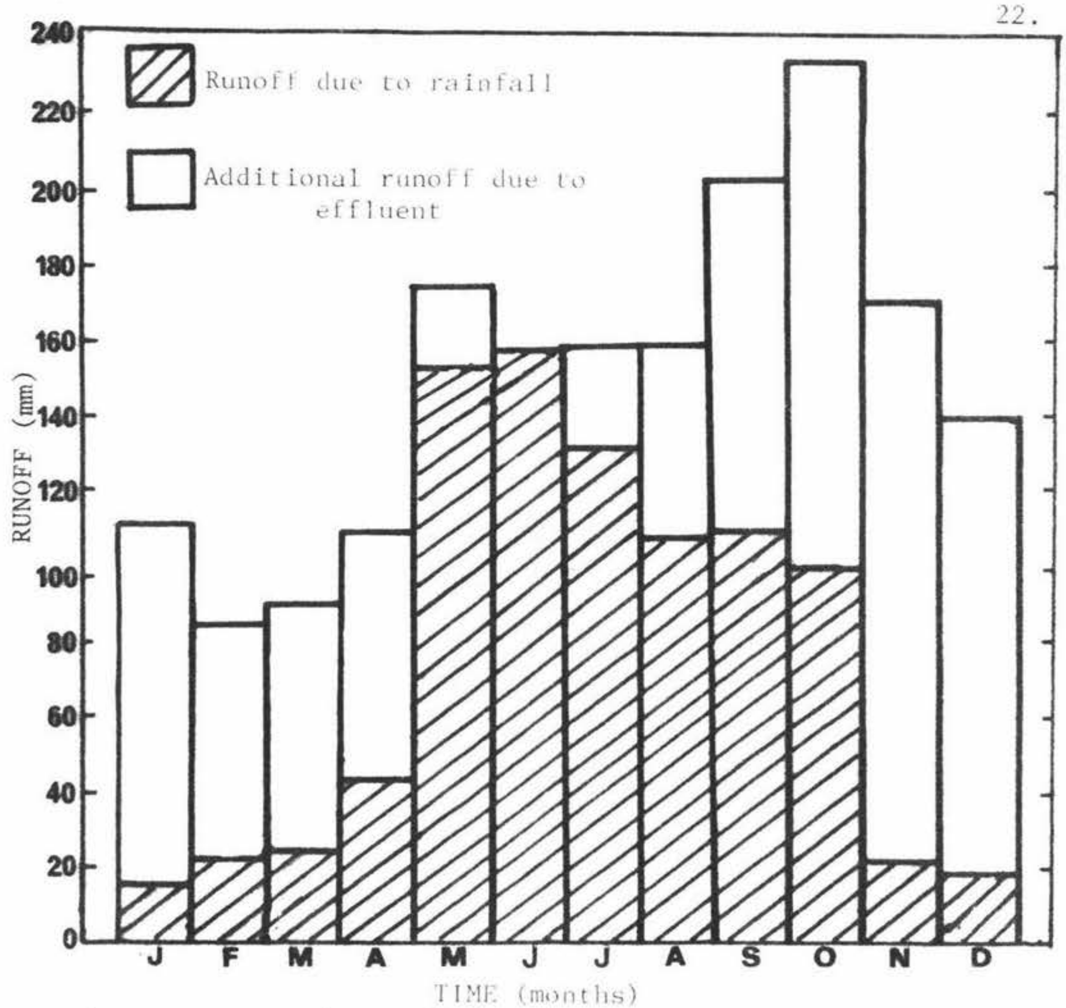

Fig. 4.1 Average seasonal runoff for the Te Rehunga site in the presence and absence of the irrigation component.

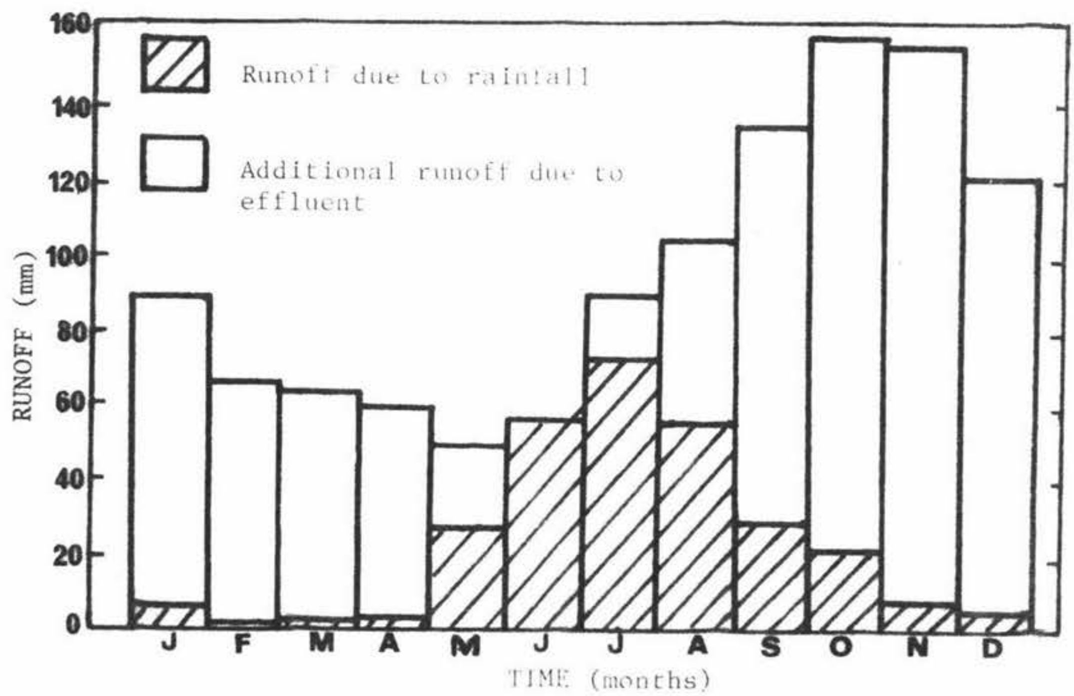

Fig. 4.2 Average seasonal runoff for the Longburn site in the presence and absence of the irrigation component.

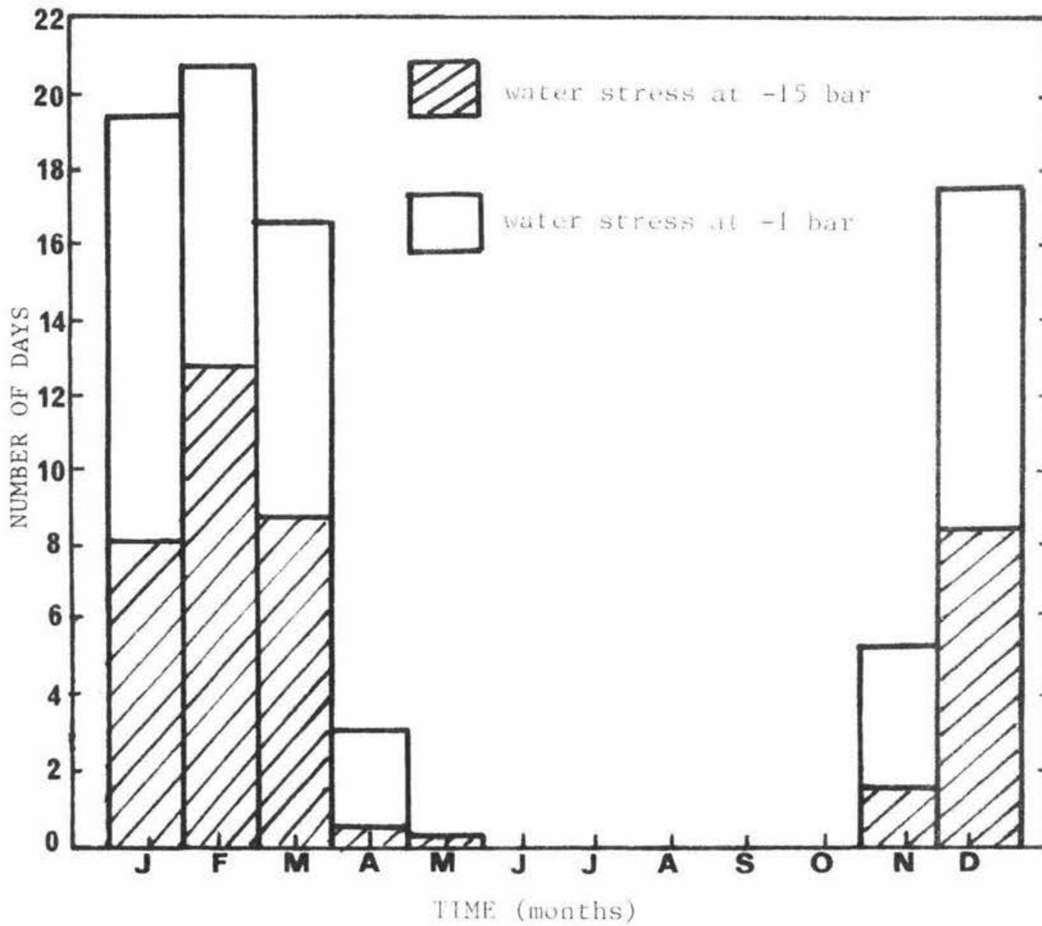

Fig. 4.3 The average number of days per month on which the Te Rehunga site is likely to be water-stressed (at the -15 bar and -1 bar levels) in the absence of irrigation.

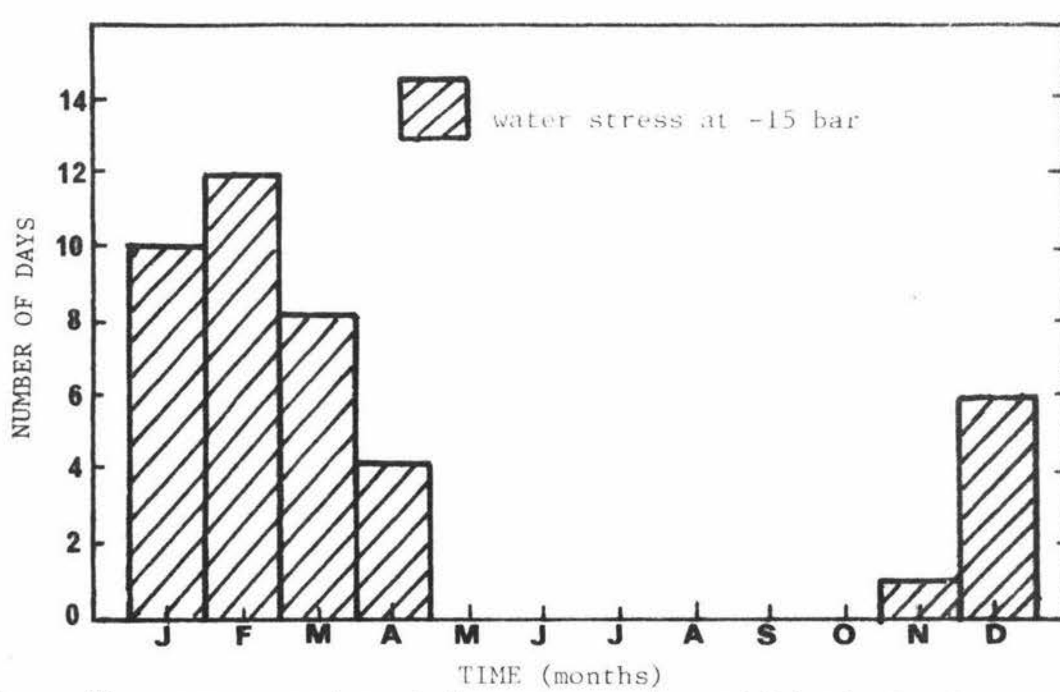

Fig. 4.4 The average number of days per month on which the Longburn site is likely to be water-stressed (at the -15 bar level) in the absence of irrigation.

These results suggest that permanent wilting point conditions would occur on average for approximately forty days per year over the November to April period. The -1 bar results from Fig. 4.3 indicate, however, that plants may be water stressed for a much greater length of time; the Te Rehunga site results show that an irrigation advantage may be obtained on approximately eighty days per year. The predicted stress periods at Longburn show a remarkably similar pattern to those at Te Rehunga. It would appear therefore that the shallow rooting habit at Te Rehunga is compensated for by the higher rainfall.

It is noted from Table A2.3, that the lower limit of readily available water (-1 bar) at the Te Rehunga site will occur when the soil is at a water deficit of -43mm. Using this value and the evapotranspiration data in Table A2.2, it is possible to calculate the maximum allowable return interval for the irrigation, if crop water stress is to be avoided. For the December-January period (with an evapotranspiration rate of around 4mm day^{-1}), the maximum return interval for the Te Rehunga site, if no rainfall occurs, is approximately eleven days. This interval is likely to be significantly longer (approximately nineteen days) for the Longburn and Tokomaru sites, however, because of the greater estimated rooting depth (Table A2.3).

A further point noted from the water balance data pertains to the likelihood of surface runoff. By adding the maximum rainfall intensity to the effluent application rate an estimate may be obtained for the maximum hydraulic loading rate likely to occur at a disposal site. Relating these estimated values to the measured soil infiltration rates may then indicate whether or not surface runoff will occur.

Maximum hourly rainfall intensities are approximately an order of magnitude higher than the maximum daily figures (Gilman, 1964). So in order to estimate the maximum hourly intensities, maximum daily rainfall values in Table A2.5 were raised by a factor of ten.

Thus the estimated maximum hourly rainfall intensities likely to have occurred at Te Rehunga and Longburn are equivalent to 1700mm day^{-1} and 920mm day^{-1}, respectively. Adding these values to the mean effluent application rate of 30mm $hour^{-1}$, the estimated maximum hydraulic application rates likely to occur are found to be equivalent to 2500mm day^{-1} and 1640mm day^{-1} for the Te Rehunga and Longburn sites, respectively.

Even though the maximum estimated hydraulic loading onto the Te Rehunga site is high it is still less than the observed infiltration rates at the site (Table 5.3). It would appear, therefore, that ponding at Te Rehunga would be unlikely, unless a high groundwater table restricts infiltration. At Longburn, however, infiltration rates for some areas of the disposal site were found to be less than the maximum estimated hydraulic loading rate (Section 7.2.2), hence some surface ponding may be expected, at least in the depression areas. Since infiltration rates in the depression areas at Longburn were found to be less than 20mm day^{-1}, frequent surface ponding events would be expected (at least as frequent as the return interval).

CHAPTER FIVE

THE EFFECTS OF DAIRY FACTORY EFFLUENT ON SOIL HYDRAULIC CONDUCTIVITY

CHAPTER FIVE

THE EFFECTS OF DAIRY FACTORY EFFLUENT ON SOIL HYDRAULIC CONDUCTIVITY

5.1 Introduction

5.1.1 Reduction in soil saturated hydraulic conductivity following effluent application

A principal design factor for a municipal, industrial, or agricultural liquid waste irrigation scheme is the hydraulic loading (Canham et al., 1955; McDowall and Thomas, 1961). In determining the rate and frequency of waste application, therefore, consideration must be given to the infiltration and percolation characteristics of the site profile.

Examples have been cited where waste irrigation schemes have failed when the rate and frequency of application were based on soil saturated hydraulic conductivity (K) values recorded prior to the initial waste application (Bendixon et al., 1969; Canham 1955; Jones and Taylor, 1965). There is considerable evidence to suggest that a high percentage of such failures are associated with the blockage or clogging of the soil pore space following effluent application (Eldridge, 1947; De Vries, 1972; Rice, 1974). An understanding of this clogging process would be of considerable benefit in the design and operations of waste disposal systems. From the design viewpoint it would be advantageous if a prediction could be made prior to site construction and waste application as to the likely decrease in K after operation. Information obtained on this blockage may have implications for site selection, particularly if soils are found to differ in their behaviour following waste application. With an understanding of the clogging process, site management practices could be controlled so as to minimise any likely K decrease.

5.1.2 A review of observations made on pore clogging and of the mechanisms involved

There have been reports of dairy factory waste disposal sites failing as a result of decreased infiltration rates (United States Environmental Protection Agency, 1971). However, no information on the extent and causes of this decrease is available. Because of this shortage of literature on dairy waste clogging of soil pore space it is necessary to examine related research carried out on food processing wastes, municipal sewage effluent and animal wastes in order to obtain the background for this study. The major aspects of the clogging process to be examined include: the extent of blockage, the conditions under which clogging is observed, the zone in the soil profile where the blockage is situated and the likely mechanisms involved in clogging.

Jones and Taylor (1965) examined the effects of sewage effluent on artificially-packed sand columns. Their results showed that the effluent could reduce the K value of the column to less than 2% of the initial value; a complete blockage was never observed however. The decrease in K recorded by Jones and Taylor happened three times more quickly under anaerobic conditions than in an aerated environment. The most rapid decrease in K was observed to occur with the smallest particle size material exposed to a non-aerated environment. Thomas et al. (1966) also reported a more rapid K decline in non-aerated columns compared with the aerated columns. De Vries (1972) observed the occurrence of clogging under a continuous application of domestic wastewater, but no drop in K when a twenty-two hour rest period between applications was allowed. McCalla (1950) was able to induce a decrease in K to a value less than 10% of the initial K by applying a 1% glucose solution to soil columns. A subsequent recovery of K was observed within a few days of this decrease.

From an experiment in which cattle feedlot runoff water was applied to "undisturbed" soil cores, Lehman and Nolan-Clark (1975) observed a decline in K from 50mm $hour^{-1}$ to 0.06mm $hour^{-1}$ over a twenty day period; a field trial using the same waste material produced similar results. Another field trial designed to study the effect of wastewater on soil K was undertaken by Bendixon et al., (1969). In this trial cannery wastes were observed to cause a gradual decline in K during the spraying season. This decline was found to be temporary, however, since K values increased between the spraying seasons.

It appears, therefore, as though wastewater application to a soil can markedly reduce K. The rate and extent of this reduction would be determined by a number of factors, the most significant of which appear to be the type of waste material and the environmental conditions prevailing around the time of application.

Three mechanisms have been proposed as being the possible causes of K decline following waste application to soils. Firstly, there is a chemical clogging mechanism. Wastes of high alkalinity, particularly with high sodium levels, have been known to cause a decline in K by dispersion of soil colloids (Wilcox, 1949). To avoid this dispersion Wilcox recommended that the sodium content of irrigation water should not constitute more than 80% of the soluble mineral ions present, and that the total concentration of cations be less than 25mg l^{-1}. McDowall and Thomas (1961) reported that sodium as a percentage of the total cations for milk wastes is under 17%, thus indicating that chemical clogging would be unlikely. Secondly, there is a biological blockage process. Addition of readily oxidisable organic wastes to a soil generally induces a rapid increase in both the number and activity of soil bacteria (McCalla, 1950). Most researchers consider the increased amount of bacterial polysaccharide material associated with this higher level of microbial activity to be the main cause of biological clogging (Rice, 1974), although gas production resulting from microbial activity has also been suggested as a clogging mechanism (Pillsbury and Appleman, 1945). The third proposed mechanism of clogging is a physical blockage process resulting from suspended material in the effluent blocking the soil conducting pores.

Most of the research into clogging following wastewater application to a soil has been aimed at identifying the blockage mechanisms involved. To achieve this aim workers have found it necessary to employ experimental techniques which isolate the biological from the physical blockage processes; the use of biological inhibitors or low temperature treatments appear to be the most effective techniques.

Johnson (1958) observed that chlorination slowed down the rate of K decline in continuously submerged soil and sand columns. Allison (1947), using mercuric chloride and ethylene oxide gas to control microbial activity, showed that the reduction in hydraulic conductivity in soils under prolonged submergence was due to a biological process. McCalla (1950) noted that the pore blockage observed following sucrose addition to soil columns was biological since no blockage was observed at low temperatures. Work by Avnimelech and Nevo (1964) has provided exceptionally useful information on the biological blockage process. Their work involved measuring the rapidity and extent of K decline following application of different waste material to sand columns. Low carbon to nitrogen (C:N) ratio wastes were found to induce a rapid but short-lived K decline. Higher C:N ratio wastes appeared to take longer to induce a K decrease, but the lowered K value remained for as long as one month after the application. An excellent correlation was noted between the extent of K decline and the polysaccharide or polyuronide levels in the columns. Similar findings were also noted by Mitchell and Nevo (1964). Results from the above research indicate that it may be possible to predict the rate and extent of K decrease following waste application by referring to the C:N ratio and stability of the added organic material.

Any immediate decline in K following effluent application would implicate a physical blockage mechanism. Rice (1974) observed a marked K decline in sand columns within minutes following an application of high suspended solids effluent. Rice concluded from this experiment that the most important factor in preventing a K decline with in-soil wastewater disposal systems was to maintain a low suspended solids level of the effluent. Further research demonstrating the physical blockage process was undertaken by Daniel and Bouma (1974). This research involved application of domestic sewage effluent to "undisturbed" soil cores taken from a low permeability silt loam. The results showed that the extent of K decline depended to a large extent on the shapes and sizes of the suspended solids in the effluent.

Most research has shown the clogged zone to be situated at or near the soil surface. Rice (1974) found that by removing the top 40mm of a clogged sand column the initial K value could be almost completely restored; any blockage occurring deeper than 40mm was attributed to entrapped gases. Lehman and Nolan-Clark (1975) found the clogging from cattle feedlot runoff also to be at the soil surface as removal of the top 20mm restored the initial K value. Using tensiometers spaced at regular intervals Jones and Taylor (1965) were able to demonstrate that the zone of blockage in a sand column overlain by gravels was at the gravel-sand interface; this result being the only cited exception to blockage occurring at the soil surface. Since the effect of wastewater application on soil K appears to be largely confined to the surface layer, it would seem reasonable to assume that useful information on the blockage process could be obtained solely with the use of surface layer soil cores.

5.1.3 The relevance of previous work to clogging from dairy factory effluent

From effluent analysis data (McDowall and Thomas, 1961; Marshall, 1975) it is evident that dairy factory wastes can have high levels of both dissolved and suspended organic solids. Undiluted cheese and casein whey has a total solids level of 6%, approximately a quarter of which is suspended material. Although this concentrated whey is diluted (by approximately 10:1 on average) before irrigation onto pasture it would still represent a high organic loading. Marshall (1975) found the biological oxygen demand (BOD) of casein washwater to be in the range of 6700 to 9300 mg l^{-1}, it was noted however that this high BOD wastewater would be mixed, approximately on a 50:50 basis, with the floor and machinery washings before spray irrigation. From a survey of all dairy factories in the United States, the United States Environmental Protection Agency (1971) found the mean BOD value of effluent to be 2500 mg l^{-1}. These values for dairy factory effluent are markedly greater than the 200 to 400 mg l^{-1} range normally cited for secondary sewage effluent (McDowall and Thomas) and are comparable to values obtained by Sanborn (1953) and Pound and Crites (1973) for food processing wastes.

Marshall (1975) found the level of suspended solids in effluent from one New Zealand dairy factory to range from 120 mg l^{-1} to 1500 mg l^{-1} with a mean value of 460 mg l^{-1}. A similar range of values was noted by Pound and Crites (1973) following a survey of a number of dairy factories in the United States. Once again, this level appears to be substantially higher than is commonly found in secondary sewage effluent where a range of between 10 and 100 mg l^{-1} is normally cited (Pound and Crites), and is comparable to values for food processing wastes. Cattle feedlot runoff can register very high suspended solids levels; a mean value of 3000 mg l^{-1} was recorded by Lehman and Nolan-Clark (1975) in their study, thus explaining the rapid and extensive K decline which was observed.

With the relatively high organic and suspended solids levels in dairy factory effluent, it could be expected that the biological and physical clogging mechanisms occurring with other effluent types would also be present when dairy factory wastewater is applied to soil. Of the research cited into effluent blockage of pore space, the work of McCalla (1950) and Avnimelech and Nevo (1964) is of particular relevance to dairy factory effluent studies, since the substrates employed would be similar in nature to that in dairy factory wastewater. However, since Avnimelech and Nevo's results illustrate that the soil's reaction to wastewater application varies with effluent composition, even the data obtained using these related substrates must be regarded with some caution.

Another problem faced in interpreting much of the data on effluent clogging of soil space stems from the use of artificially-packed columns for the study. Such a practice does have advantages, particularly in allowing a direct comparison to be made between treatments (since all columns would exhibit relatively similar initial K values), but serious disadvantages are recognised. The hydraulic flow pattern through artificially-packed soil may be entirely different to the flow pattern in the undisturbed soil (Williams and Allman, 1969). Because there may be this marked discrepancy in flow pattern, particularly for soils in which flow is dominated by continuous macropores, the validity of results obtained from artifically-packed columns is questionable. The use of "undisturbed" cores is therefore preferable in laboratory studies of effluent clogging. Even results obtained from such studies may still be of limited use to the field situation however, since laboratory conditions are unlikely to be identical to the field conditions; this is particularly relevant to information obtained on the rates and extent of K decline.

In conclusion, therefore, it appears as though data obtained from municipal, industrial and agricultural effluent research may be used to indicate the likely effects of dairy factory wastes on soil K. However, the limitations with interpreting such data must be realised. Problems also exist when relating laboratory data to the field situation, particularly when artificially packed soil columns have been used.

5.1.4 Increases in soil hydraulic conductivity

With any blockage process occurring in a soil there must either be an associated recovery process or else the soil will tend towards zero K in time. Case studies where no long term K change was observed suggest that a balance may exist between the blockage and recovery processes. This balance may alter in time, however, giving seasonal fluctuations in soil K (Bendixen et al., 1969; Linderman and Stegman, 1971). The possibility of no flow channel blockage at a disposal site is unlikely. Even if blockage due to the effluent itself is discounted, mechanisms such as soil slaking (Ehlers, 1975), surface smearing and compaction from stock, machinery and rainfall (Pound and Crites, 1973; Pearson et al., 1975) and root growth in the pore space (Barley, 1954) cannot be avoided completely.

Few researchers have examined the mechanisms inducing soil K increases. From the little information which is available, however, it appears as though a number of mechanisms may cause a K increase. These include the removal of material obstructing an existing flow channel, new channel formation by soil organisms, crack formation during drying of the soil surface, and root decay or dessication. The blockage removal process may be differentiated from the other three mechanisms since it is a "recovery process", whereas the remaining mechanisms result in the formation of new flow channels. The two most likely means by which a blockage may be removed from an existing pore would be by soil organism activity or physical dislodging as liquid percolates through the soil profile.

Of the channel forming organisms in New Zealand soils, earthworms are the group most likely to induce marked increases in soil K. Ehlers (1976) showed in a field experiment that infiltration values of a soil could be increased significantly within a few months as a result of earthworm activity. From his studies Ehlers concluded that the large diameter pores resulting from earthworm activity have a profound influence on water movement through soil.

The mechanism of channel formation by cracking would be of limited importance in this study since the soil profile at a liquid waste disposal site would be unlikely to dry out to the point at which cracking occurred.

Possibly the only in-depth study into the effects of plant roots on K was an experiment conducted by Barley (1954). In this experiment, Barley measured the effect of root growth and decay on the permeability of synthetic sandy loam cores. He found that the root decay process resulted in the freeing of soil water conducting channels, hence causing an increase in K values of the cores.

It appears as though research into the mechanisms responsible for K increase in soils may be warranted for two major reasons. The first of these reflects the limited amount of literature available on the subject thus indicating that little is known about the process. The second reason relates to the balance between blockage and recovery processes, whereby it may be necessary to consider mechanisms inducing an increase in K in order to effectively examine the blockage process.

5.2 Aims of Permeability Studies

Laboratory studies using intact cores comprise a large section of the work described in this dissertation. It was considered that information obtained from these studies may enable an understanding of both the effluent blockage process and the associated recovery process. The use of soil cores to examine changes in saturated hydraulic conductivity (K) was considered preferable to field studies, largely because comparisons between standardised treatments are more easily achieved. The laboratory core studies have limitations however (Section 5.4.9), and it is necessary to use field data to complement the laboratory observations.

5.3 Experimental Design

In order to compare the reaction of different soils to an application of dairy factory effluent, it was necessary to adopt a standard experimental procedure. This standard procedure involved applying a 35mm equivalent (150 ml) fluid dose (either water or effluent) to cores on every second day, and the removal of any fluid remaining on the core surface after three hours of ponding. This procedure was initially adopted since it was considered to be a realistic representation of a disposal site situation while still ensuring adequate time for the measurement of K. From preliminary studies using cores treated with water only, it was found that this applied water regime resulted in a stable K value after an "equilibrium period". Since this stable K value was generally attained by the fourth fluid application, an equilibrium period of at least eight days was employed prior to any effluent application. The K value at the start of treatment was thus relatively stable and is termed the "equilibrium K".

Selection of cores for each treatment was made at the end of the equilibrium period. With the large variability observed in equilibrium K values for cores from the same soil group, it was evident that comparison between treatments would be difficult. Thus "core pairings" were used, whereby two similar equilibrium K value cores were allocated to the comparative treatments.

Measurement of K was made during the period of ponding. Because preliminary studies revealed that K may vary markedly throughout the ponding period, it was necessary to standardise the time at which measurements were made. The procedure adopted was to measure K during the ten to twenty minute period after percolation through the core commenced.

5.4 Results and Discussion

Using the basic procedure described, five groups of experiments were set up to examine the effluent blockage process. In addition to these five experiments, a further laboratory study was made examining the recovery process. As a means of providing supporting evidence for the laboratory studies, two field experiments were also conducted.

Most of the results obtained from the "undisturbed" soil core studies are presented in graphical form, showing changes in K with time. Because of the large variability between the K values of individual cores it was not practical to use the same scale for all graphs. Instead, soil cores with similar K values are presented in groups. Results from the "undisturbed" laboratory core studies are also expressed on the basis of the percentage change in K in relation to the "equilibrium K".

Results from the field experiments are presented in tabular form. These results have been statistically analysed using the "students t test" (Wetherill, 1967). Tables show the mean value ($\bar{x}$), the standard deviation (s) and the number of samples used (n). The level of significance is represented as follows: n.s. indicates no significant difference between the means, * indicates significance at the 90% confidence level, ** significance at the 95% level and *** significance at the 99% level.

5.4.1 "Undisturbed" soil core reaction to repetitive applications of simulated whey effluent

The repetitive application experiment was designed to provide a general indication of soil reaction to a standard effluent application. Preparation of the simulated effluent used in the experiment is described in Section 3.4.4.

Six 150 ml doses, each equivalent to a 35 mm application of the simulated whey effluent, were applied to cores on days 0, 2, 4, 6, 10 and 14, where day '0' coincided with the attainment of "equilibrium K". From day 6 onward, a four day interval between fluid applications was adopted. This extended interval, a modification on the standard design procedure, was introduced to enable a comparison to be made between a two and four day spelling interval. Cores from five soils were used.

These were Tokomaru silt loam (TSL) 0-100mm depth, Tokomaru silt loam 100-200mm depth, Kopua silt loam (KSL) 0-100mm depth, Kopua silt loam 100-200mm depth, and Manawatu silt loam (MSL) 0-100mm depth.

The results from these studies are shown in Fig. 5.1-5.10. An examination of the K changes in the seventeen cores receiving the simulated whey effluent shows that individual soil cores vary markedly in their reaction to the effluent application. Common features can be recognised from the results however and it is these features which form the basis of the following discussion.

The application of effluent does appear to induce a marked decline in K within two days. Of the seventeen effluent treatment cores, fifteen exhibited a substantial decrease in K by day 2. The magnitude of this initial two day decline varied considerably, with cores of high "equilibrium K" values generally showing the greatest absolute decrease in K. On a proportional change basis, however, no observable differences were noted between the high and low K value cores. Since both high and low K value cores registered K decline following effluent application, it appears as though blockage due to the effluent is possible over a range of conducting pore sizes.

The mean percentage decrease in K for all soils as a result of the initial effluent application was 46% compared with a mean decrease over the same period of less than 5% for the water only treatment cores.

Extending the interval between applications from two to four days resulted in a variable response, and it is not possible to identify any common features or to draw any firm conclusions from the data. Measurement of evaporation from a free water surface in the laboratory showed losses of approximately 2mm day^{-1}. It is possible therefore that the marked K increase in cores 1A, 1B, 1J and 10 upon introduction of the four-day spelling interval may to some extent be due to partial drying and incipient cracking in the core or to dessication of the blockage material.

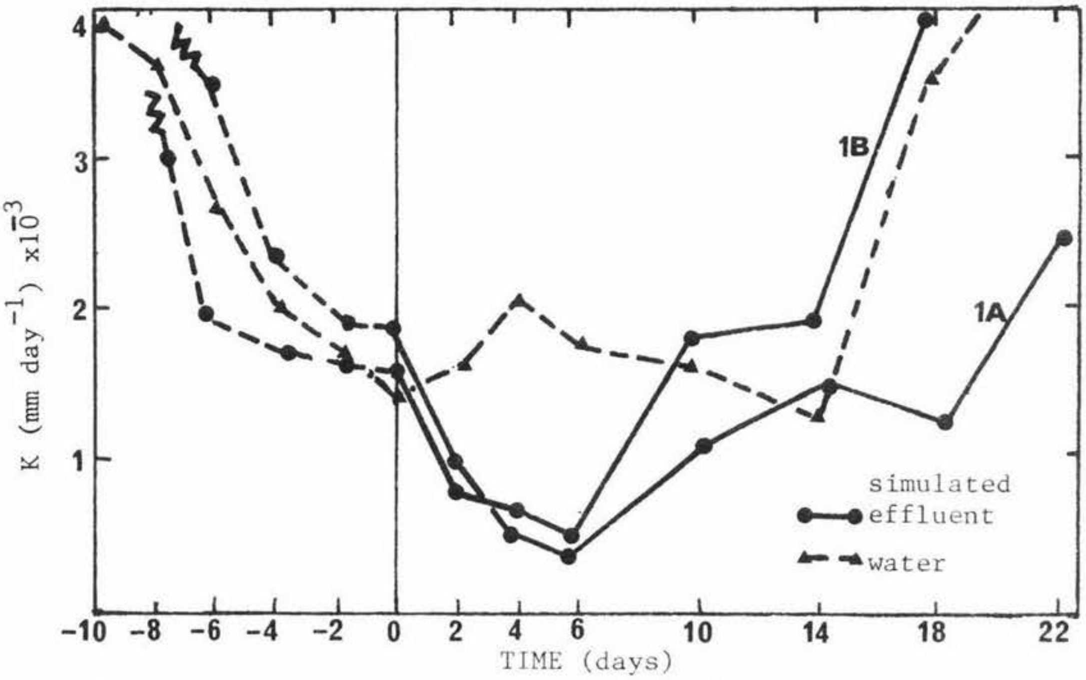

Fig. 5.1 Changes of K with time following repetitive applications of simulated effluent to two Tokomaru silt loam (0-100mm) cores, with a water only treatment core for comparison.

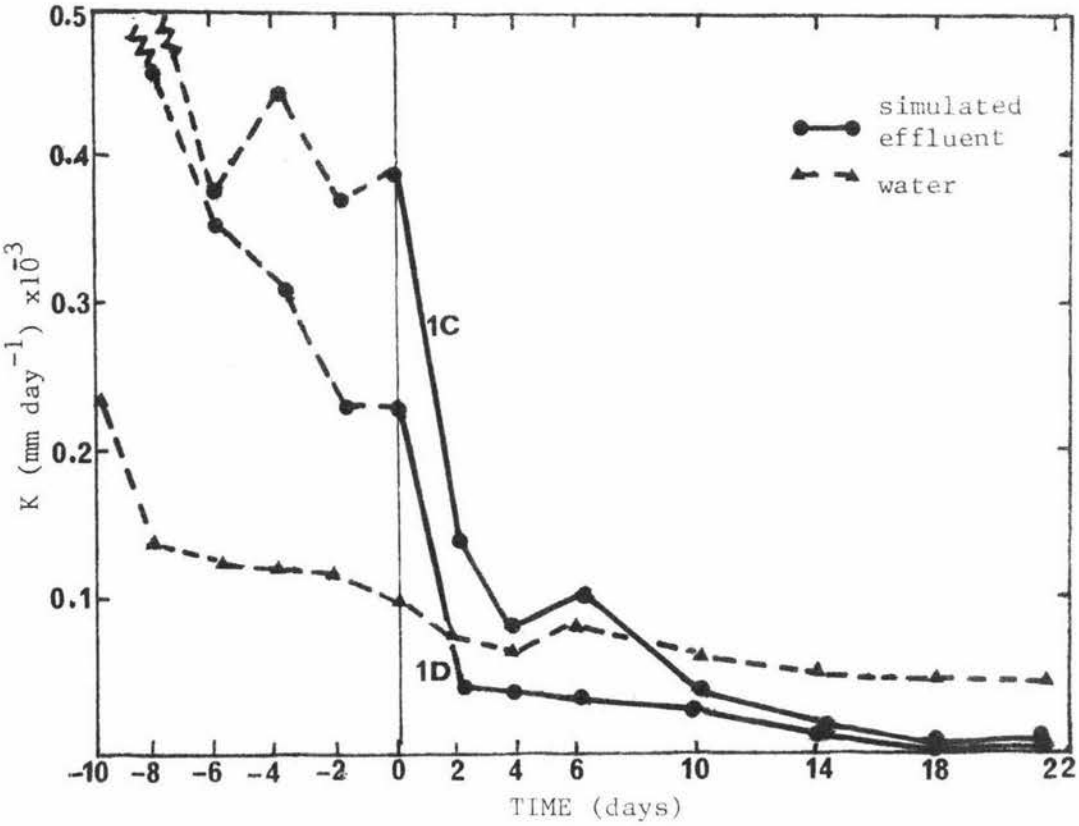

Fig. 5.2 Changes of K with time following repetitive applications of simulated effluent to two Tokomaru silt loam (0-100mm) cores, with a water only treatment core for comparison.

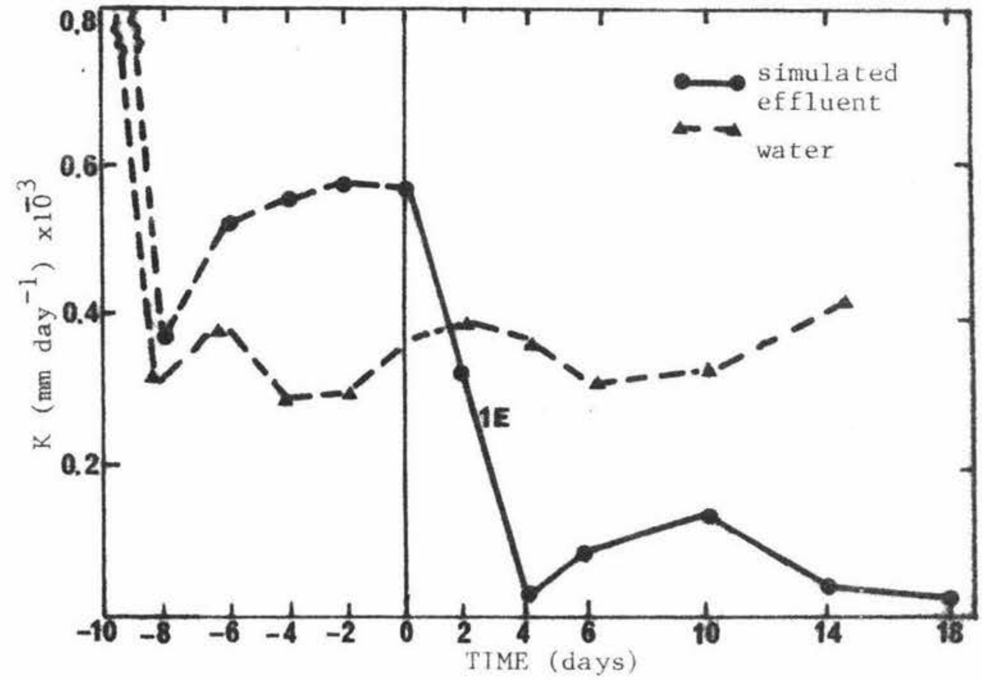

Fig. 5.3 Changes of K with time following repetitive applications of simulated effluent to a Tokomaru silt loam (100-200mm) core, with a water only treatment for comparison.

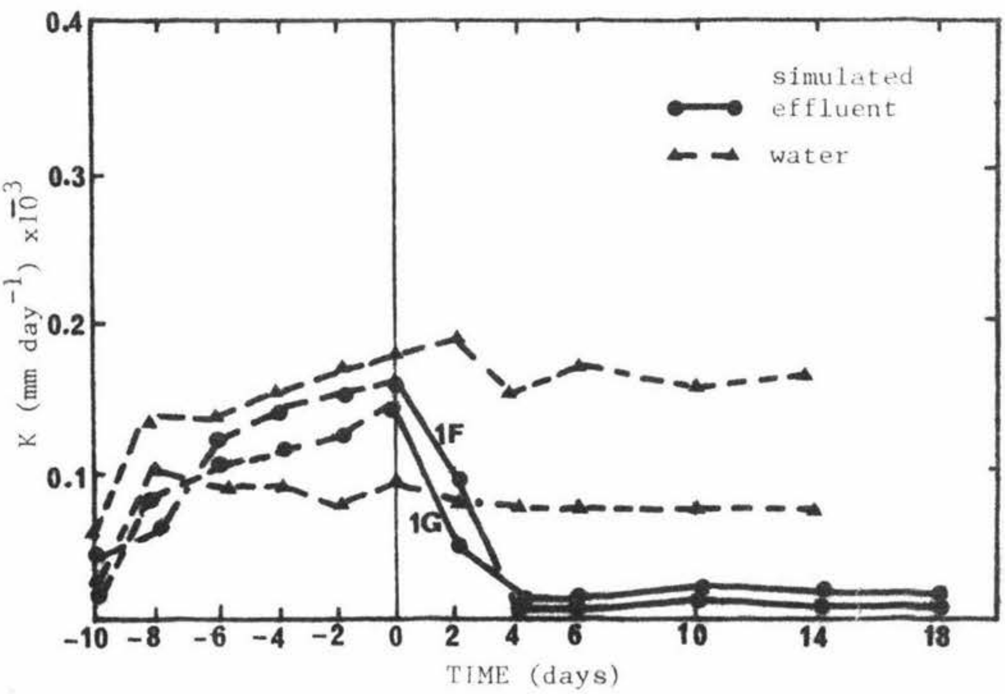

Fig. 5.4 Changes of K with time following repetitive applications of simulated effluent to two Tokomaru silt loam (100-200mm) cores, with two water only treatment cores for comparison.

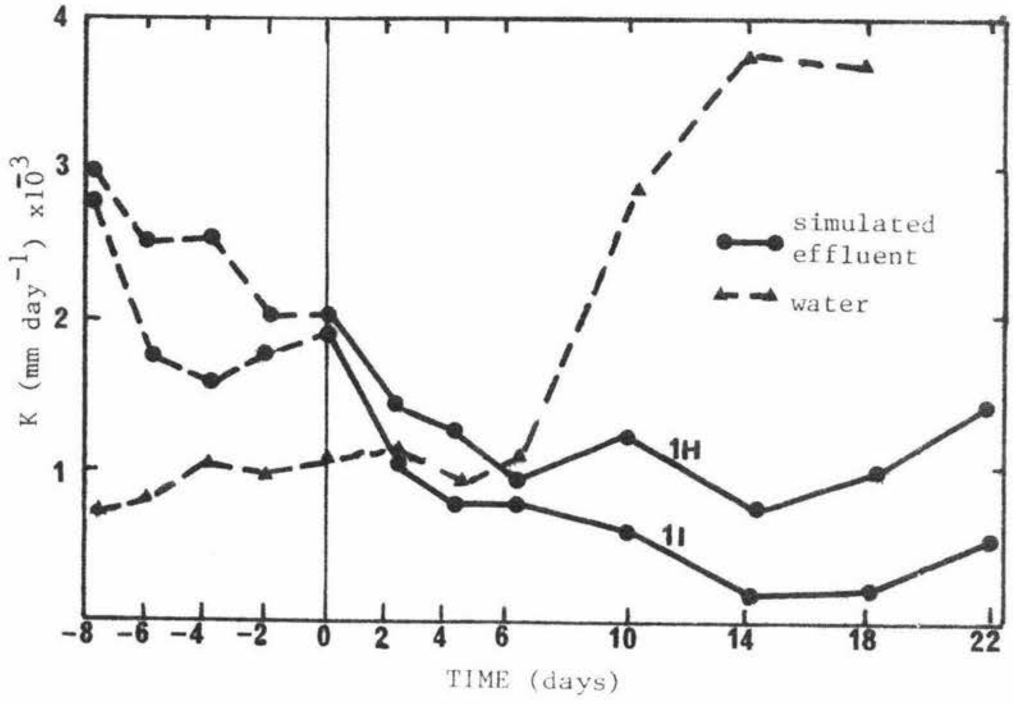

Fig. 5.5 Changes of K with time following repetitive applications of simulated effluent to two Kopua silt loam (0-100mm) cores, with a water only treatment core for comparison.

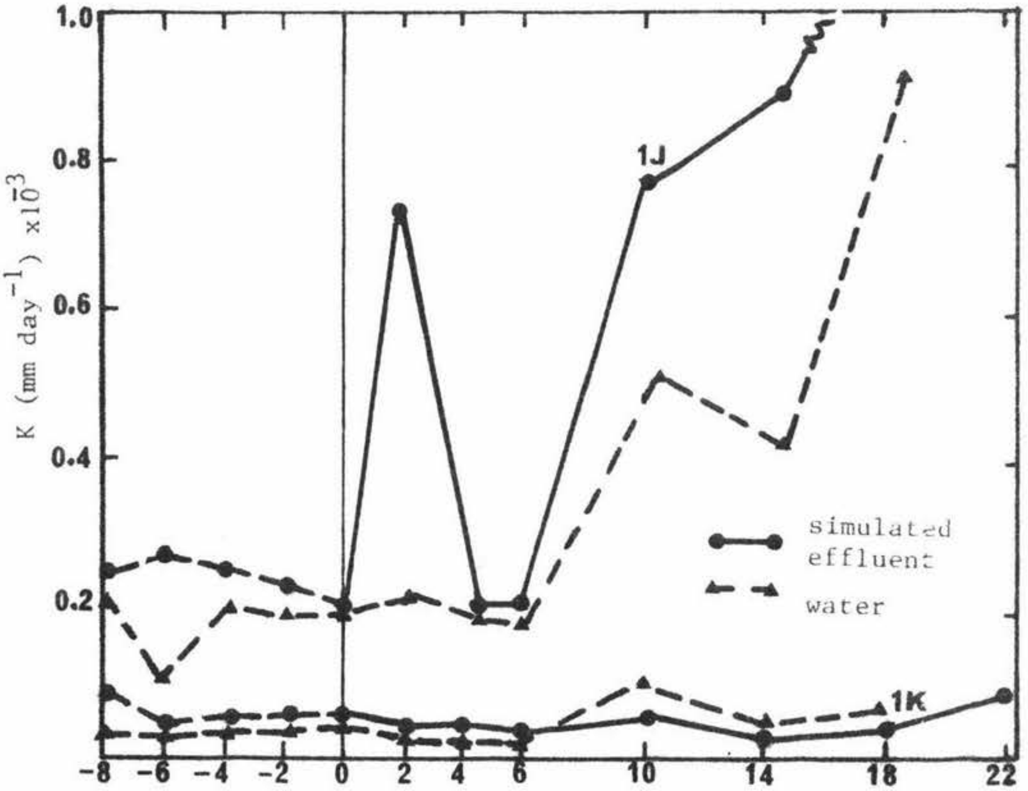

Fig. 5.6 Changes of K with time following repetitive applications of simulated effluent to two Kopua silt loam (0-100mm) cores, with two water only treatment cores for comparison.

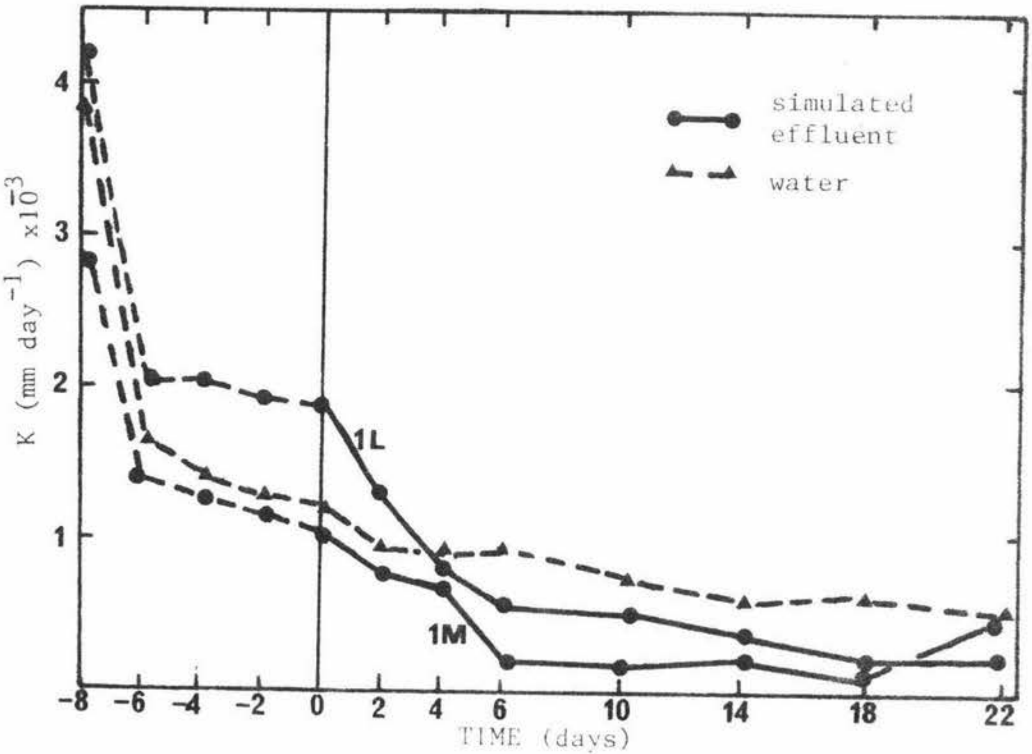

Fig. 5.7 Changes of K with time following repetitive applications of simulated effluent to two Kopua silt loam (100-200mm) cores, with a water only treatment core for comparison.

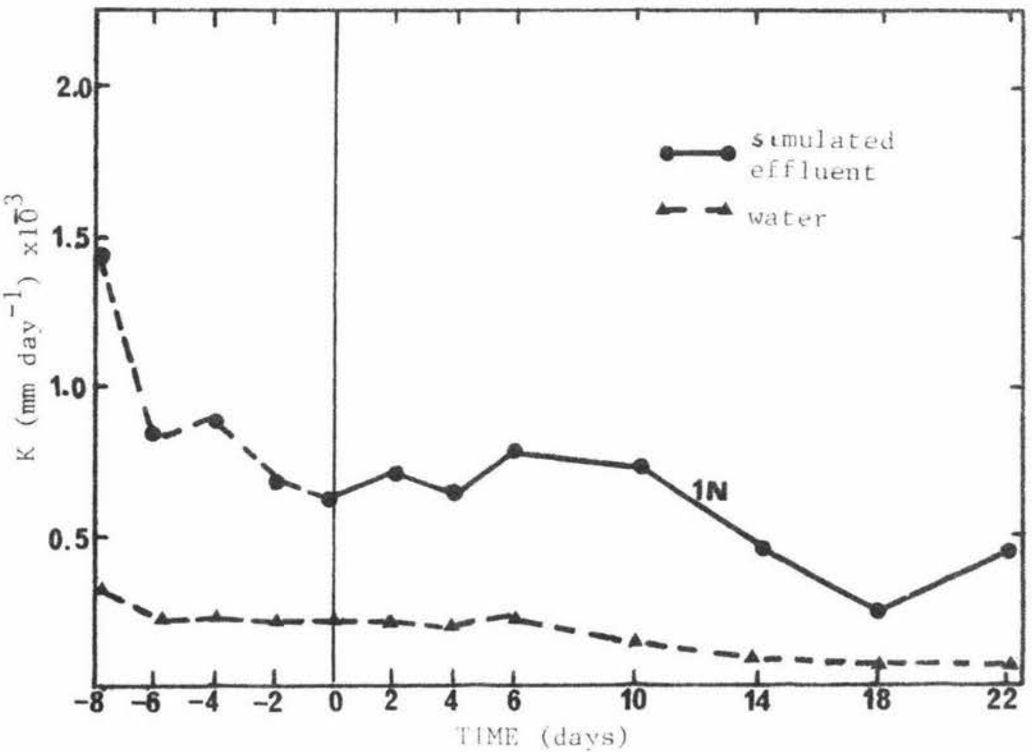

Fig. 5.8 Changes of K with time following repetitive applications of simulated effluent to a Kopua silt loam (100-200mm) core, with a water only treatment core for comparison.

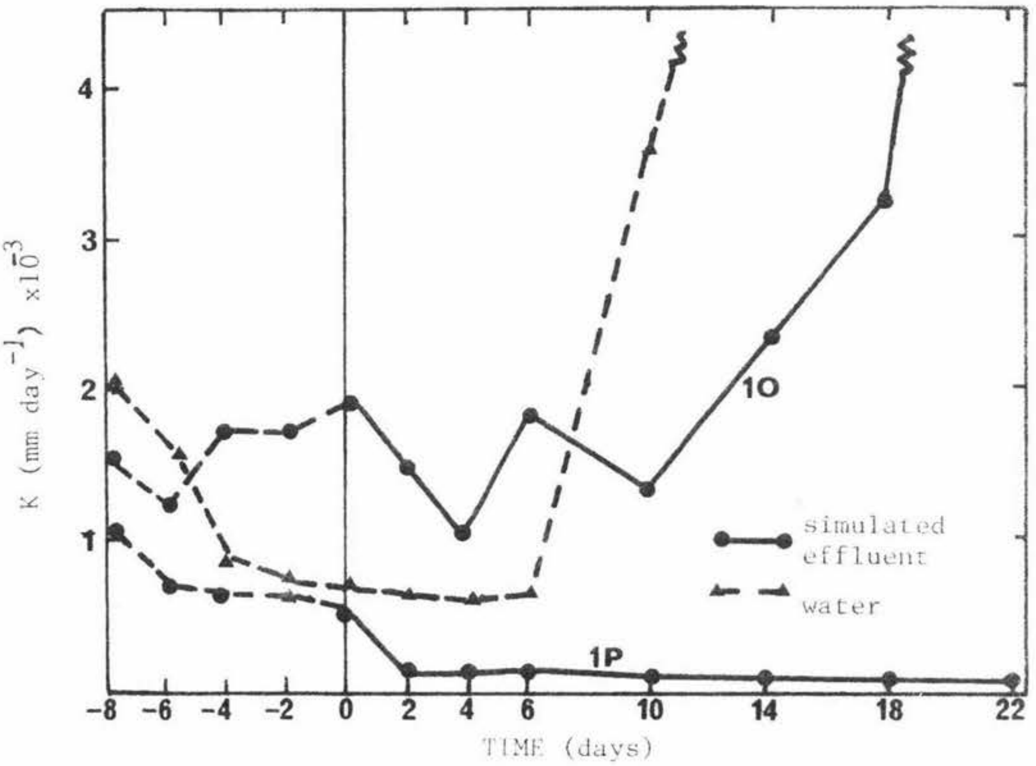

Fig. 5.9 Changes of K with time following repetitive applications of simulated effluent to two Manawatu silt loam (0-100mm) cores, with a water only treatment core for comparison.

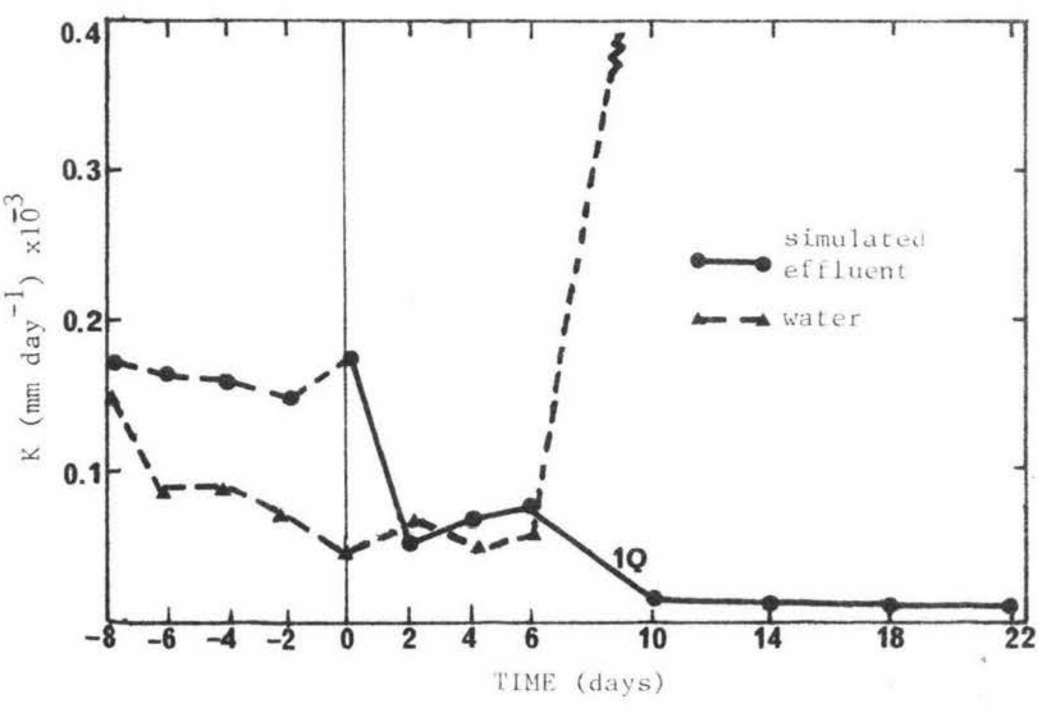

Fig. 5.10 Changes of K with time following repetitive applications of simulated effluent to a Manawatu silt loam (0-100mm) core, with a water only treatment core for comparison.

Examination of the overall change in K following repetitive effluent application shows that thirteen of the seventeen effluent treatment cores registered a marked K decline averaging around 80%, with six cores (1C, 1D, 1E, 1F, 1G and 1P) showing a decrease of approximately 99%. The marked increases in K for the remaining four cores (1A, 1B, 1J and 1O) indicate that new flow channels had been formed in these cores.

It is evident from comparisons of K changes on a soil group basis that soils differ markedly in their behaviour. Tokomaru silt loam (100-200mm) was the most susceptible of the five soils to blockage, with a mean K change over the treatment period of -99%. Low K value cores of TSL (0-100mm) and MSL (0-100mm) also appeared to be relatively susceptible to blockage, but the high K cores from these soils did not show significant overall K decreases. It would appear therefore that high K value cores may be less susceptible to long-term effluent blockage.

Kopua silt loam cores showed relatively little tendency to become blocked by the effluent. Although the effluent treatment cores for this soil did exhibit a steady decline of K with time, much of this appeared to result from factors unrelated to the effluent, since the water treatment cores exhibited a similar decline. The most likely explanation of this is slaking of soil aggregates since the soil is weakly structured and very friable.

Although K values of some cores were greatly reduced following repetitive effluent application, complete blockage, or zero K, was never induced. This fact indicates that either some conducting channels were not completely blocked, or that water movement occurred through the blockage material.

The K reduction in the sub-surface TSL cores (Fig. 5.3 and 5.4) appeared to be a relatively long-lasting phenomenon, since virtually no recovery of K was evident, even up to twenty days following the final effluent application. In comparison, K increases were seen in KSL cores and in the high conductivity TSL and MSL cores within four days of the final effluent application.

5.4.2 Changes in saturated hydraulic conductivity following a single application of simulated whey effluent to "undisturbed" cores

The single application experiment involved applying a 35mm equivalent dose of simulated whey effluent on day '0'. K was measured every second day by applying water, as described in Section 5.3. Where a second effluent application was made a minimum resting period of fourteen days was allowed. This was considered to be the time interval most representative of the spray disposal site situation, since a two week return interval is frequently used. Three soil groups, TSL (0-100mm), TSL (100-200mm) and KSL (0-100mm) were used in this experiment, and the results obtained are shown in Figs. 5.11 - 5.15.

As with the cores from the repetitive application experiment, a marked decline in K was observed within two days of the initial application. Only one of the twelve cores did not reach the minimum K value within two days, the exception, core 2C, reaching a minimum K value between day 2 and day 4.

All nine TSL effluent treatment cores showed K recovery to at least some extent. Of the three KSL cores, however, only one (2H) showed any improvement in K; with the remaining two cores (2G and 2I) actually showing further K decline. This decline was probably not a result of the effluent, however, since the water only treatment core from this soil also showed a K decrease with time. The K recovery in all nine TSL cores was initiated within eight days of effluent application, with four cores showing signs of K recovery within four days. By day 14, seven of the nine cores had exhibited essentially complete recovery to the "equilibrium K" value.

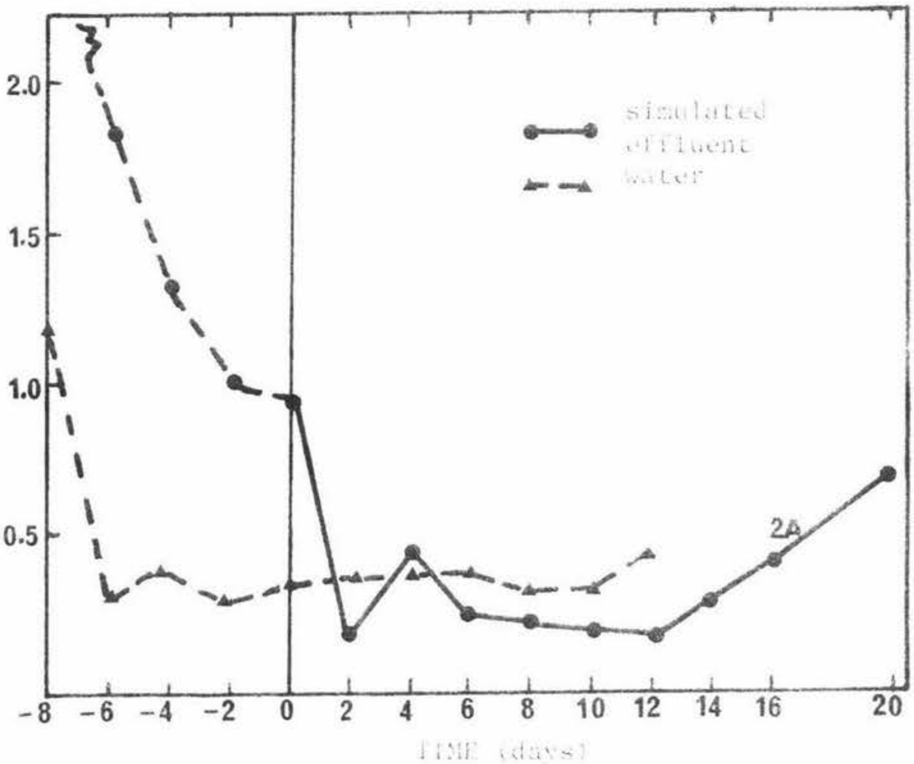

Fig. 5.11 Changes of K with time following a single application of simulated effluent to a Tokomaru silt loam (100-200mm) core, with a water only treatment core for comparison.

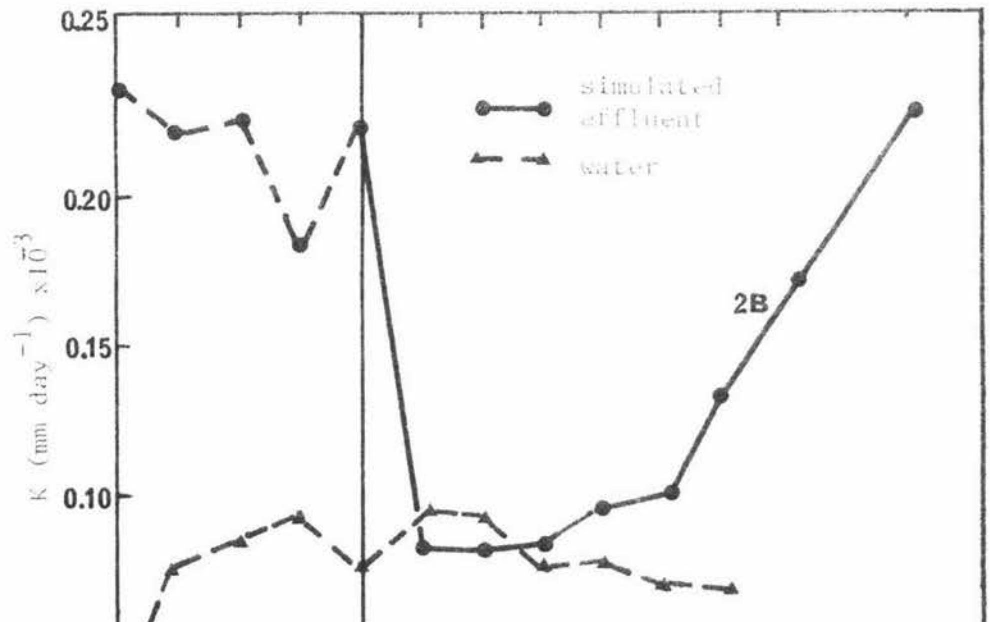

Fig. 5.12 Changes of K with time following a single application of simulated effluent to two Tokomaru silt loam (100-200mm) cores, with a water only treatment core for comparison.

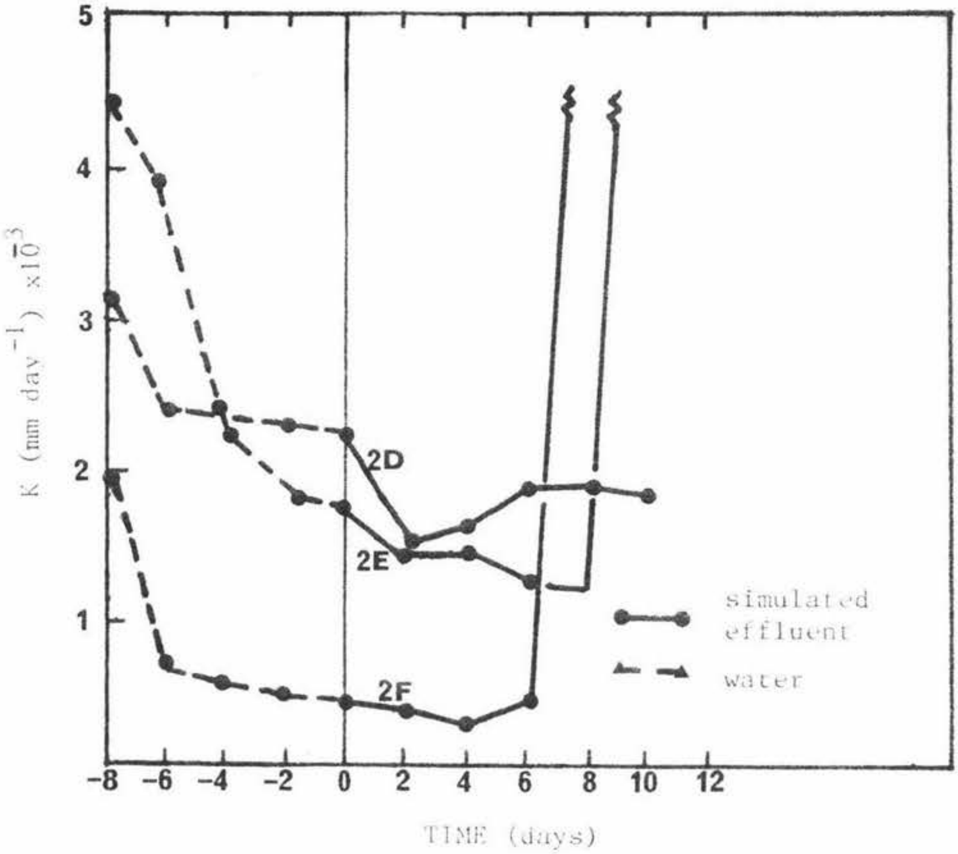

Fig. 5.13 Changes of K with time following a single application of simulated effluent to three Tokomaru silt loam (0-100mm) cores.

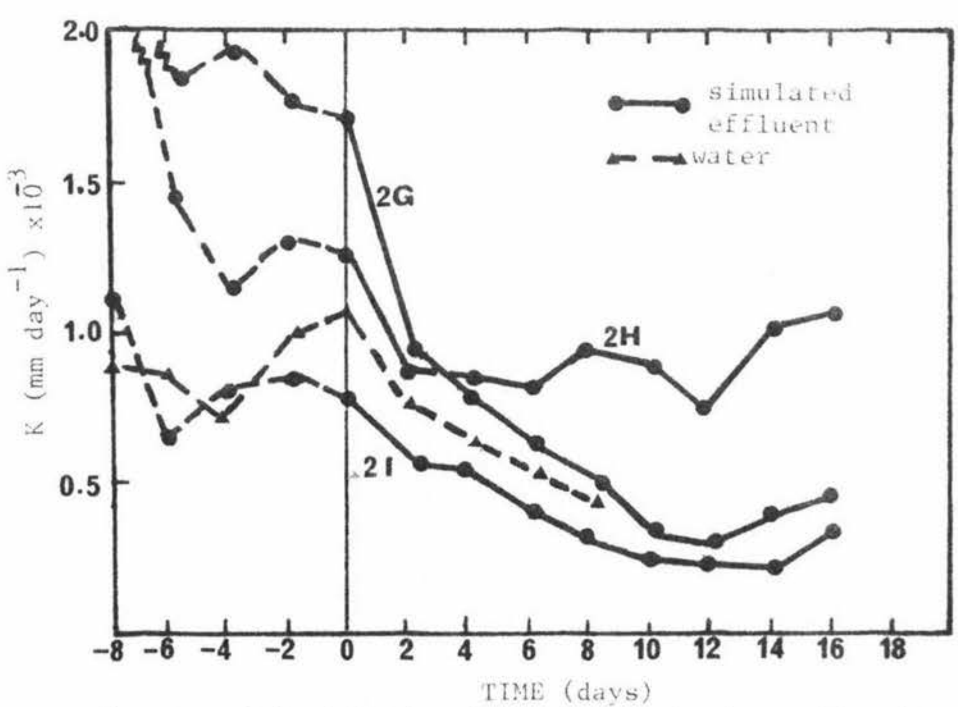

Fig. 5.14 Changes of K with time following a single application of simulated effluent to two Kopua silt loam (0-100mm) cores, with a water only treatment core for comparison.

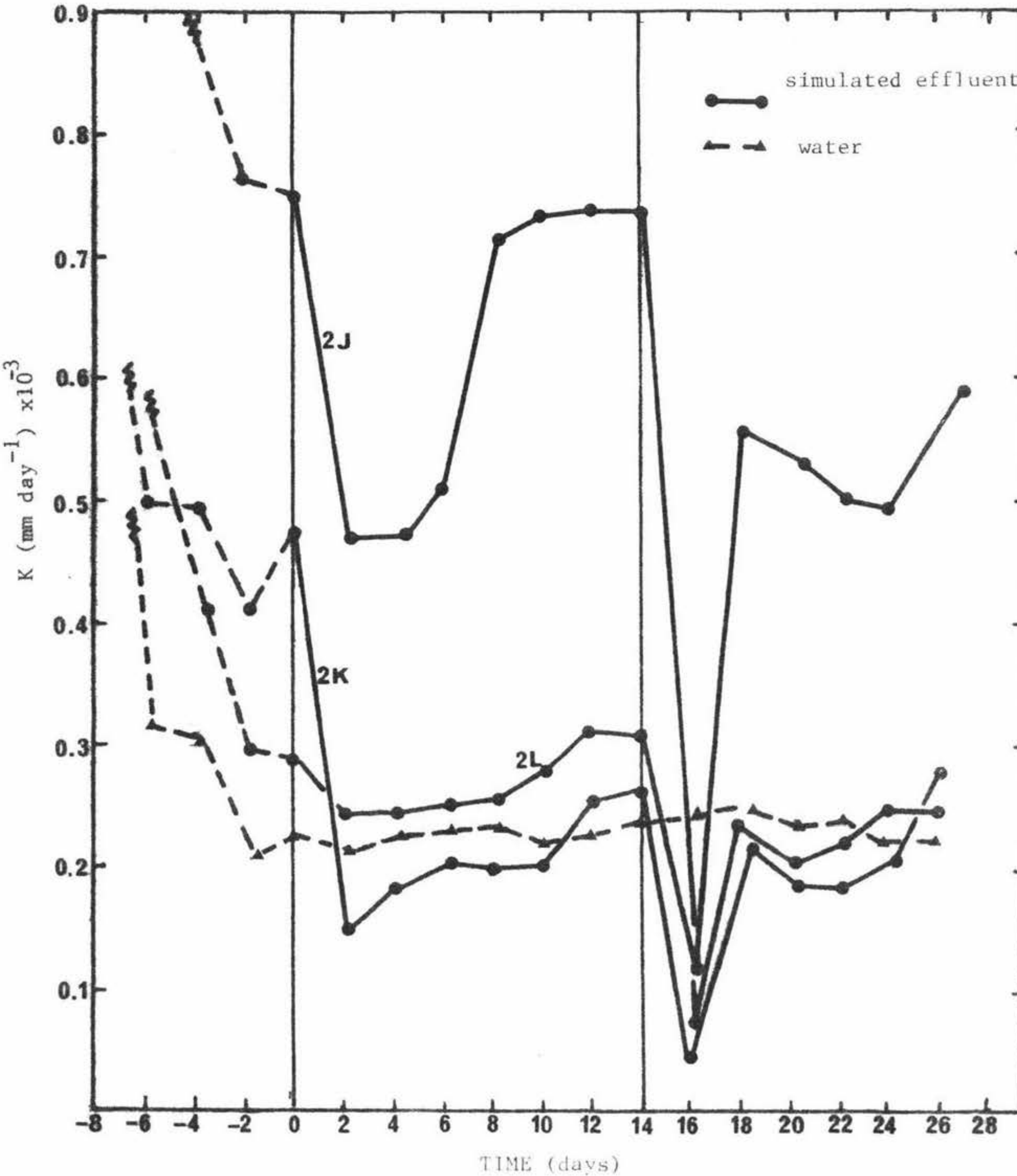

Fig. 5.15 Changes of K with time following two applications of simulated effluent (on days '0' and '14') to three Tokomaru silt loam (100-200mm) cores, with a water only treatment core for comparison.

The change in K following a second application of effluent on day 14 (Fig. 5.15) differed in two respects from the response shown to the initial application. Firstly, the extent of the decline in K was more marked following the second application; mean K decreases within two days of 40% and 88% were recorded for the initial and second application respectively. Secondly, a more rapid K recovery response was observed following the second application, although complete recovery of the "equilibrium K" for two of the three whey treatment cores was not reached within the alloted fourteen-day period. Since both the blockage and recovery response to the second effluent application is more marked than to the initial application, it appears that the soil may have become "conditioned" to the effluent. Micro-organism numbers for example, may have been built-up as a result of effluent application, thus inducing a more pronounced K change. Respirometry studies (Section 6.3.10) provide supporting evidence for the existence of a "conditioning" effect. These studies showed that the soil microbial response to the initial simulated whey application was limited, but that the response to subsequent applications was both more rapid and of greater magnitude.

Results from a modified experiment suggest that with a different moisture regime following a single effluent application it may be possible to induce a more comprehensive K decline. This experiment adopted a four-day interval between the time of effluent application and the successive water application. The K change over this four-day period was compared with the change observed using the standard two-day interval procedure. This comparison (Table 5.1) shows that a more pronounced decline was obtained with the four-day interval. Although only conjectural, it may be that the standard two-day interval is an inadequate time period for development of maximum blockage stability in subsurface Tokomaru cores, consequently after only two days the blockage material may be more readily dislodged by percolating fluid.

TABLE 5.1 Comparison of the initial changes in K between a two- or four-day spelling interval following application of simulated effluent to Tokomaru silt loam (100-200mm) cores

Treatment	Core Number	"Equilibrium K" (mm day^{-1})	K on day 4 (mm day^{-1})	Change in K (%)	Mean Change (%)
Two-day interval	1	936	187	-80	
	2	563	320	-43	-57
	3	148	53	-64	
	4	160	90	-44	
Four-day interval	5	419	9	-98	
	6	67	0.7	-99	-99
	7	160	2	-99	

5.4.3 Isolating the biological blockage mechanism

As a means of identifying any biological blockage following effluent application an experiment was designed which consisted of repetitive application treatments of either simulated effluent or simulated effluent plus inhibitor. The experimental procedure adopted was identical to that used in Section 5.4.1. Sodium azide, at a concentration of 100 mg l^{-1} of effluent solution, was employed as the microbial inhibitor. Soil groups used in this experiment were TSL (0-100 mm), TSL (100-200 mm) and MSL (0-100mm). K changes with time for all cores are shown in Figs. 5.16 - 5.21.

The decline of K with time for the effluent plus inhibitor cores was observed to be less than for the effluent treatment cores, this holding true for each "core pairing". From carbohydrate analysis of the soil column leachate (Section A4.2) it became apparent that the sodium azide used was not totally effective as a microbial inhibitor; respirometry studies (Fig. A4.1) confirmed this. Because inhibition was not absolute, the interpretation of any K changes in the inhibitor treatment cores is difficult. However, the consistent differences in the rate and extent

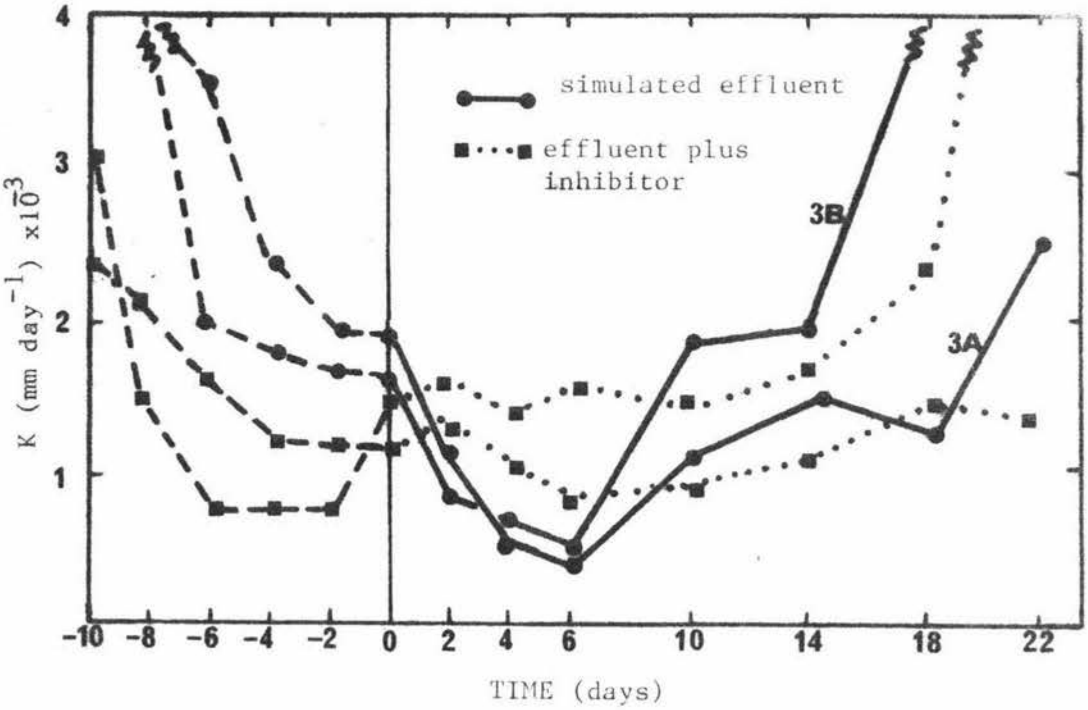

Fig. 5.16 Changes of K with time following repetitive applications of simulated effluent and effluent plus inhibitor treatments to Tokomaru silt loam (0-100mm) cores.

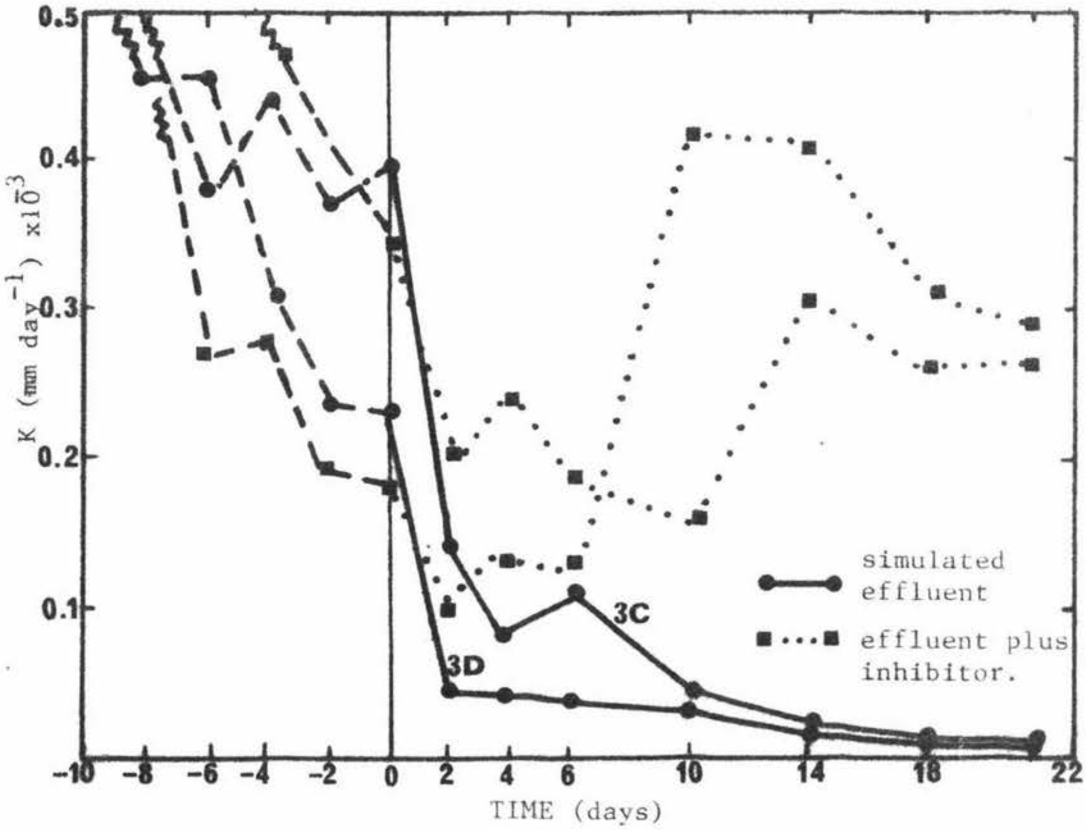

Fig. 5.17 Changes of K with time following repetitive applications of simulated effluent and effluent plus inhibitor treatments to Tokomaru silt loam (0-100mm) cores.

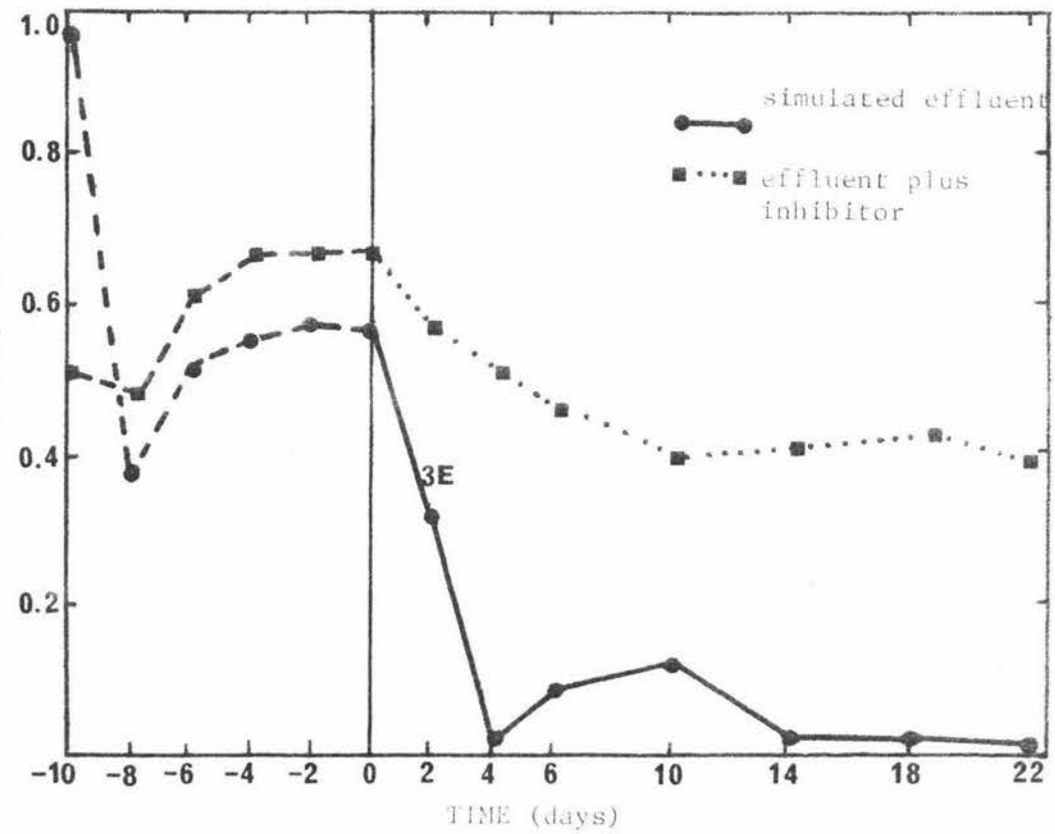

Fig. 5.18 Changes of K with time following repetitive applications of simulated effluent and effluent plus inhibitor treatments to Tokomaru silt loam (100-200mm) cores.

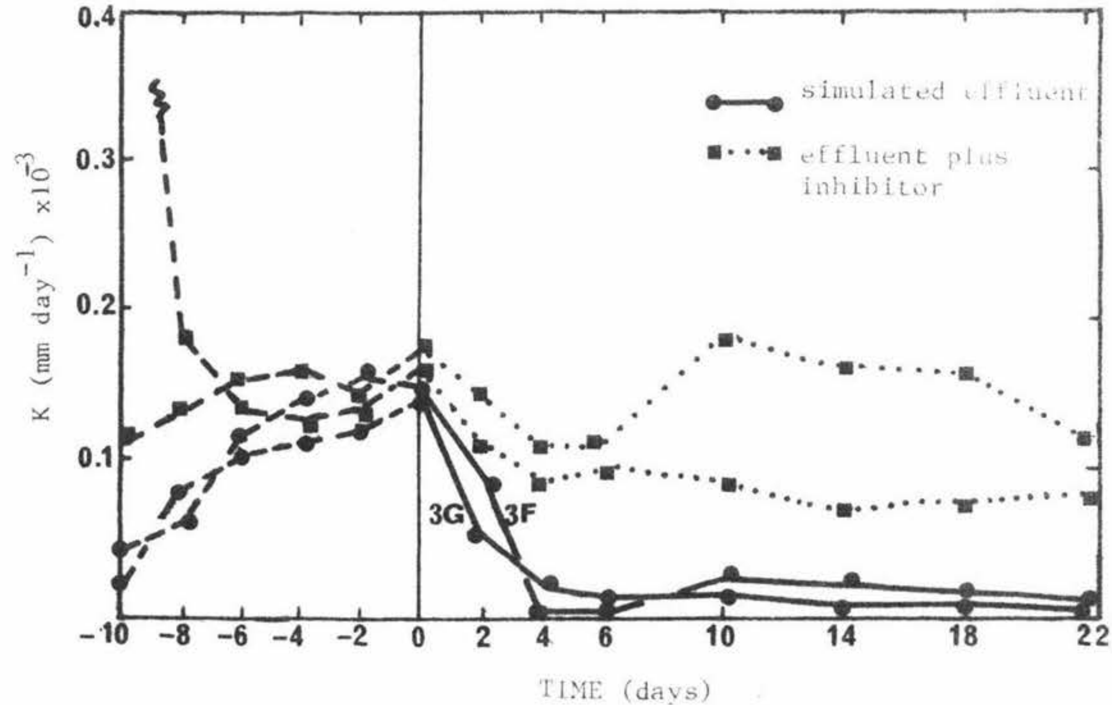

Fig. 5.18 Changes of K with time following repetitive applications of simulated effluent and effluent plus inhibitor treatments to Tokomaru silt loam (100-200mm) cores.

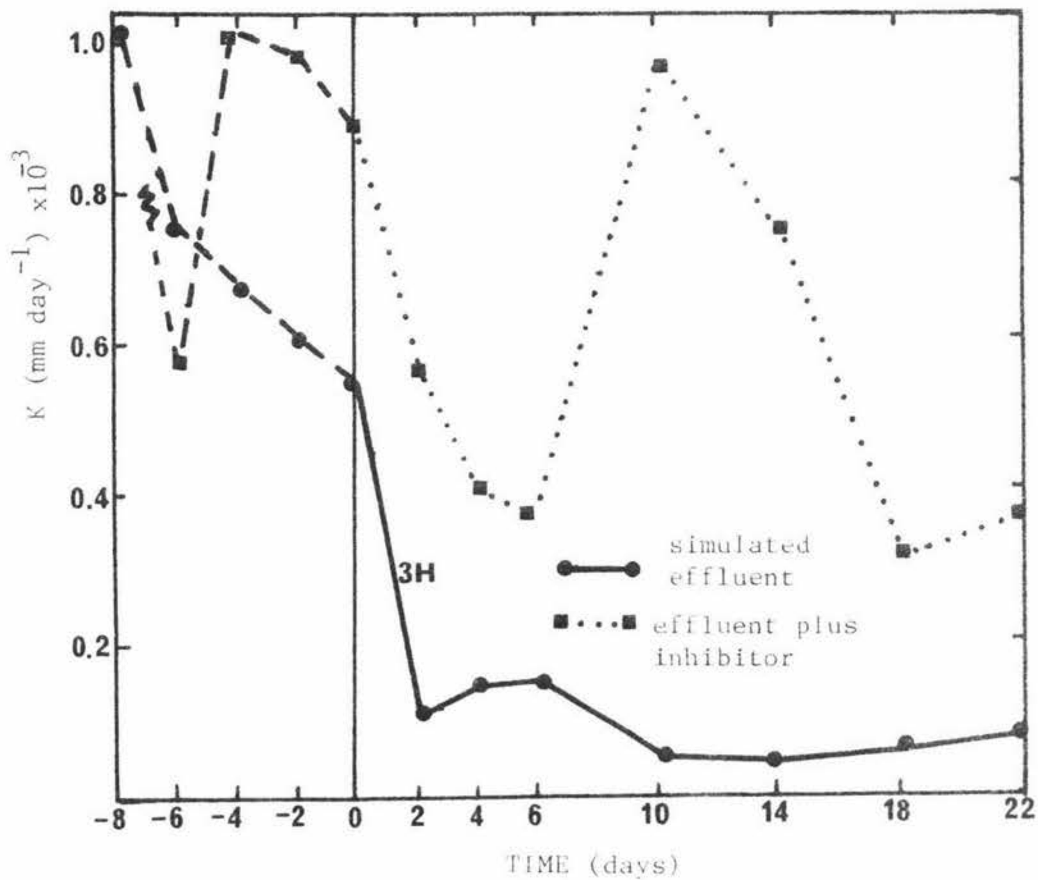

Fig. 5.20 Changes of K with time following repetitive applications of simulated effluent and effluent plus inhibitor treatments to Manawatu silt loam (0-100mm) cores.

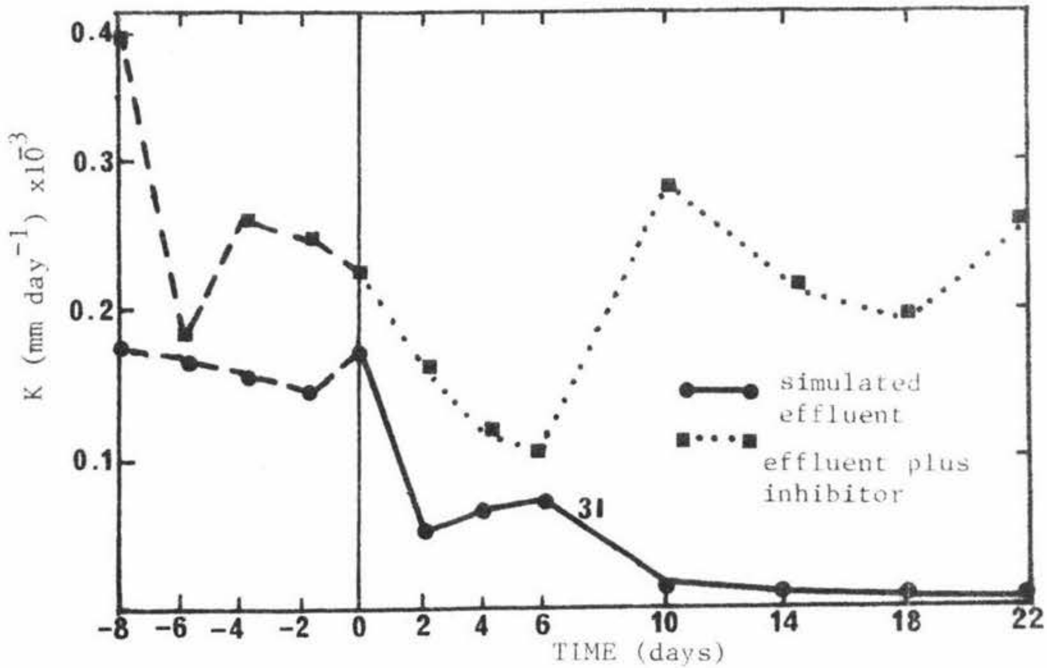

Fig. 5.21 Changes of K with time following repetitive applications of simulated effluent and effluent plus inhibitor treatments to Manawatu silt loam (0-100mm) cores.

of K change between the two treatments provides sufficient evidence to show that at least some of the blockage caused by the effluent was biologically based. Carbohydrate analyses and respirometry studies indicated that the inhibitor was most effective in the TSL (100-200mm) cores, hence results obtained from this soil would provide the best indication as to the extent of biological blockage.

5.4.4 Isolating the physical blockage mechanism

The procedure employed to identify any physical blockage following effluent application was based upon the assumption that the change in K during the 3 hour ponding period must be mainly caused by physical blockage. Following the attainment of "equilibrium K", effluent was applied to the cores and changes in K over the ponding period recorded. The sum of the physical plus biological blockages was determined by recording the K value two days after the effluent application. To ensure accurate differentiation between the physical and biological blockages K was recorded at the beginning and end of the ponding period. Three soil groups, TSL (0-100mm), TSL (100-200mm), and Karapoti silt loam (0-100mm) were used in this experiment. The results are shown in Figs. 5.22-5.25.

Of the nine effluent treatment cores used, eight registered a decline in K during the ponding period. This immediate decline was as much as 250mm day^{-1} (core 4F) with an average K change for the nine cores of -21%.

Eight of the nine cores registered a continued decline over the next two day period. The exception (core 4H), showed a marked K increase which could only be explained by the creation of a new macro-channel. The mean percentage change over the complete two day period for the eight cores exhibiting a nett decrease was -56% relative to "equilibrium K".

It appears, therefore, that the clogging observed following simulated whey application is the result of both physical and biological mechanisms. The physical blockage probably results from suspended solids in the

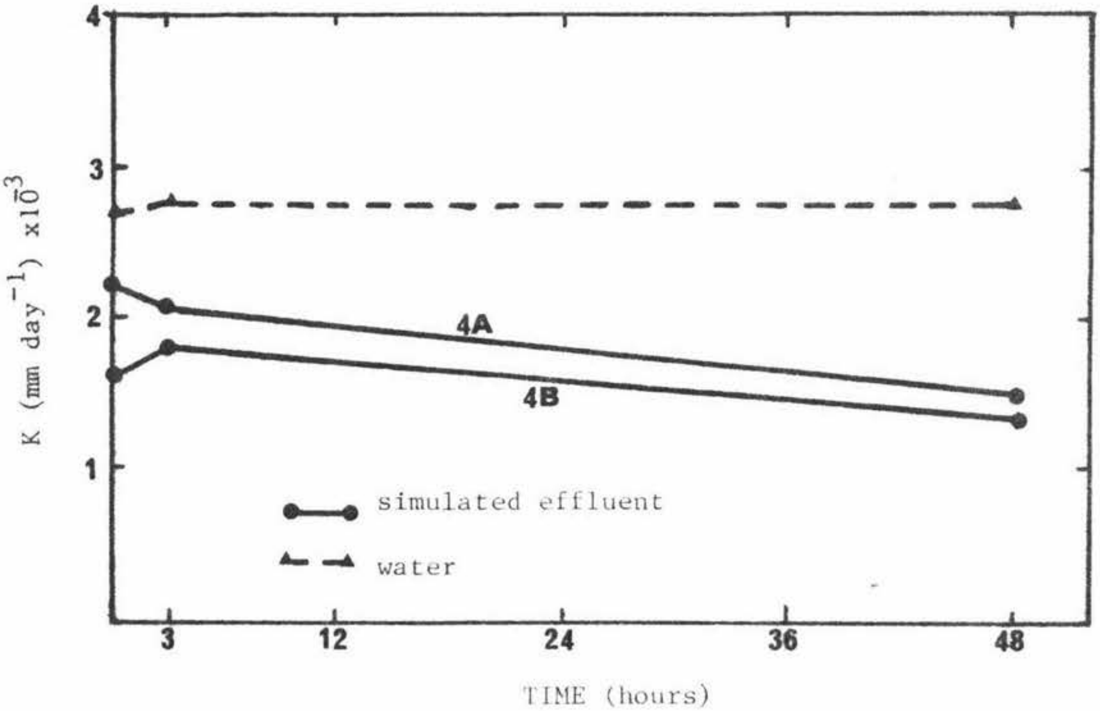

Fig. 5.22 Changes in K during ponding and over the two day period following application of simulated effluent to Tokomaru silt loam (0-100mm) cores, with a water only treatment core for comparison.

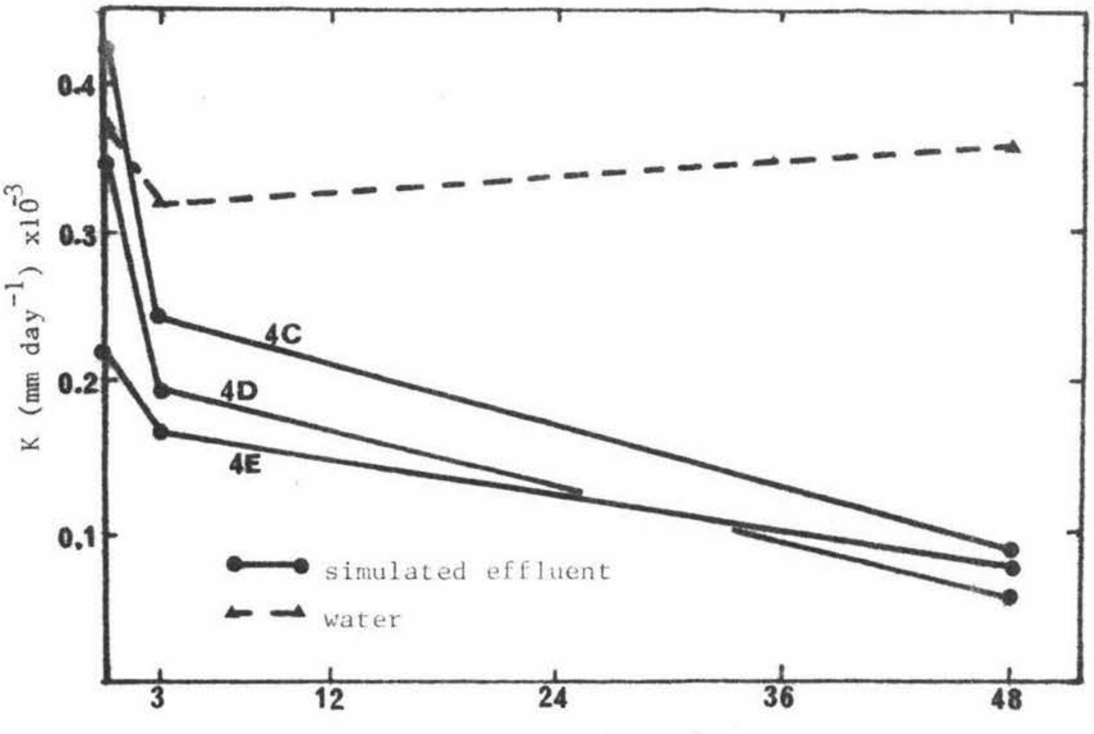

Fig. 5.23 Changes in K during ponding and over the two day period following application of simulated effluent to Tokomaru silt loam (100-200mm) cores, with a water only core for comparison.

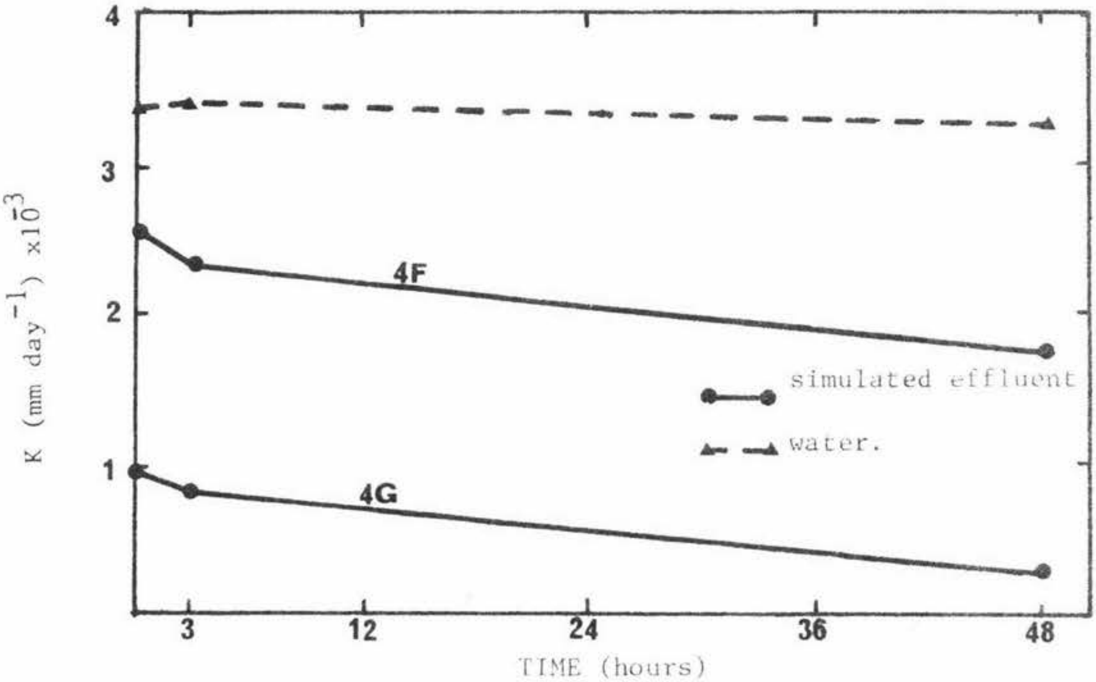

Fig. 5.24 Changes in K during ponding and over the two day period following application of simulated effluent to two Karapoti silt loam (0-100mm) cores, with a water only treatment core for comparison.

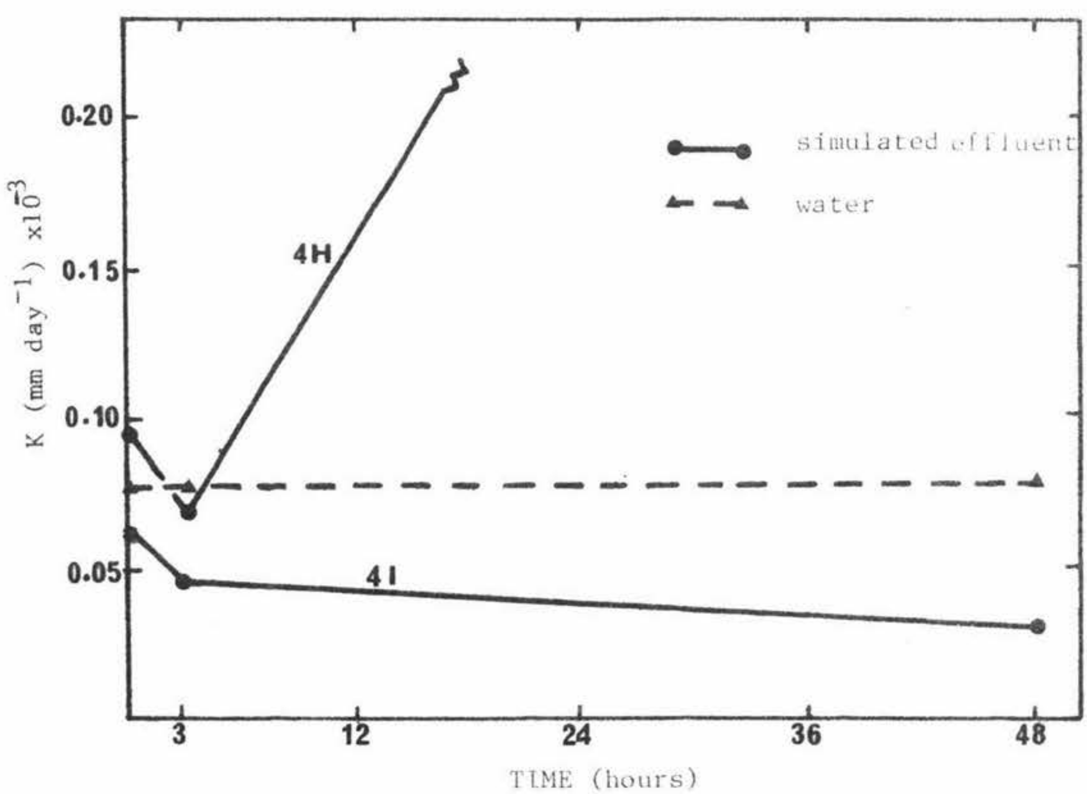

Fig. 5.25 Changes in K during ponding and over the two day period following application of simulated effluent to two Karapoti silt loam (0-100mm) cores, with a water only treatment core for comparison.

effluent becoming lodged in conducting channels. The suspended or dissolved solid material is likely to provide a substrate for soil micro-organisms, with the resulting metabolic products causing the biological blockage.

5.4.5 The extent of blockage as a function of effluent quality

The simulated whey effluent used had a total solids level of approximately 0.6%, with around 0.1% present in the suspended form. Although these values are not atypical for dairy factory effluent, spot samples taken from a local factory showed that effluent could contain suspended solids levels exceeding 0.3% (Table A5.1). Because of this an experiment was designed with the aim of comparing changes in K following application to soil cores of the simulated whey effluent and high suspended solids effluent.

Following attainment of "equilibrium K", a 35mm equivalent dose of effluent from the Longburn dairy factory containing between 0.25 - 0.35% suspended material, was applied to TSL (0-100mm) and TSL (100-200mm) cores. Immediate changes in K (during the ponding period), as well as longer term K changes, were recorded. The results obtained for the high suspended solids effluent are shown in Figs. 5.26 - 5.28. These results may be compared with the results obtained in Figs. 5.11 - 5.13 for the simulated whey effluent. A comparison of the results obtained for the two effluents shows that the high suspended solids effluent induced a more comprehensive decrease in K. The mean percentage K change in two days for the high suspended solids effluent cores was -60%, compared with a mean change of -33% for the simulated effluent. This difference was significant at the 90% level. From this result it would appear that maintaining a low suspended solids level in wastewater is an important factor in minimising any effluent blockage at a disposal site.

Figures 5.29 and 5.30 illustrate that the K change observed with the high suspended solids effluent occurred both during ponding and over the

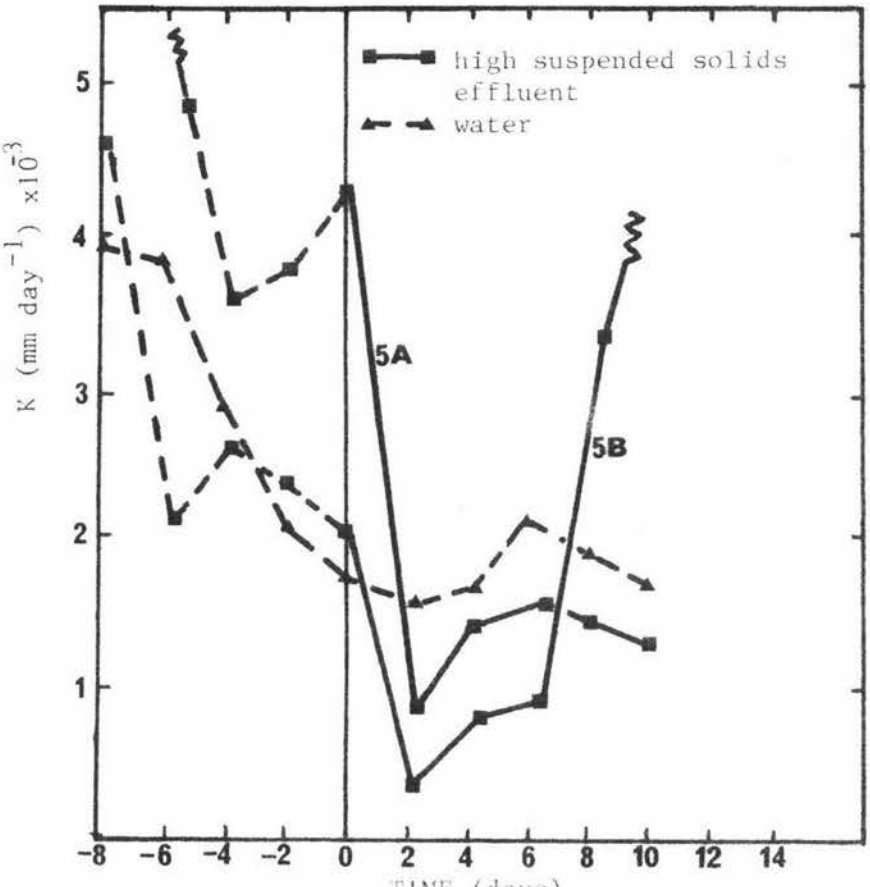

Fig. 5.26

Changes of K with time following a single application of high suspended solids effluent to two Tokomaru silt loam (0-100mm) cores, with a water only treatment core for comparison.

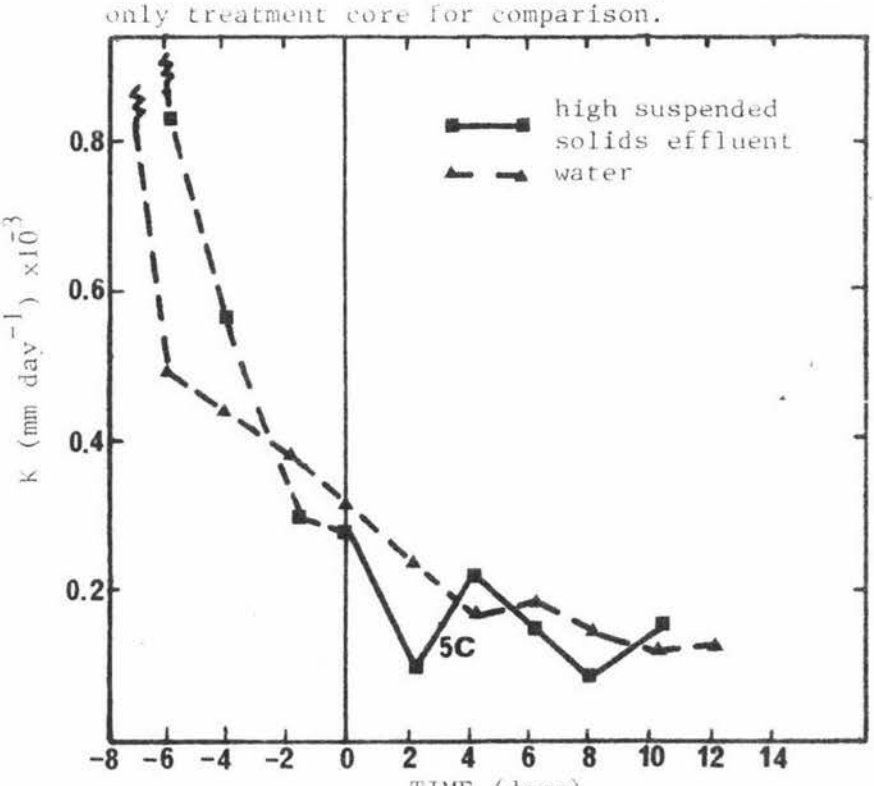

Fig. 5.27

Changes of K with time following application of high suspended solids effluent to a Tokomaru silt loam (0-100mm) core, with a water only treatment core for comparison.

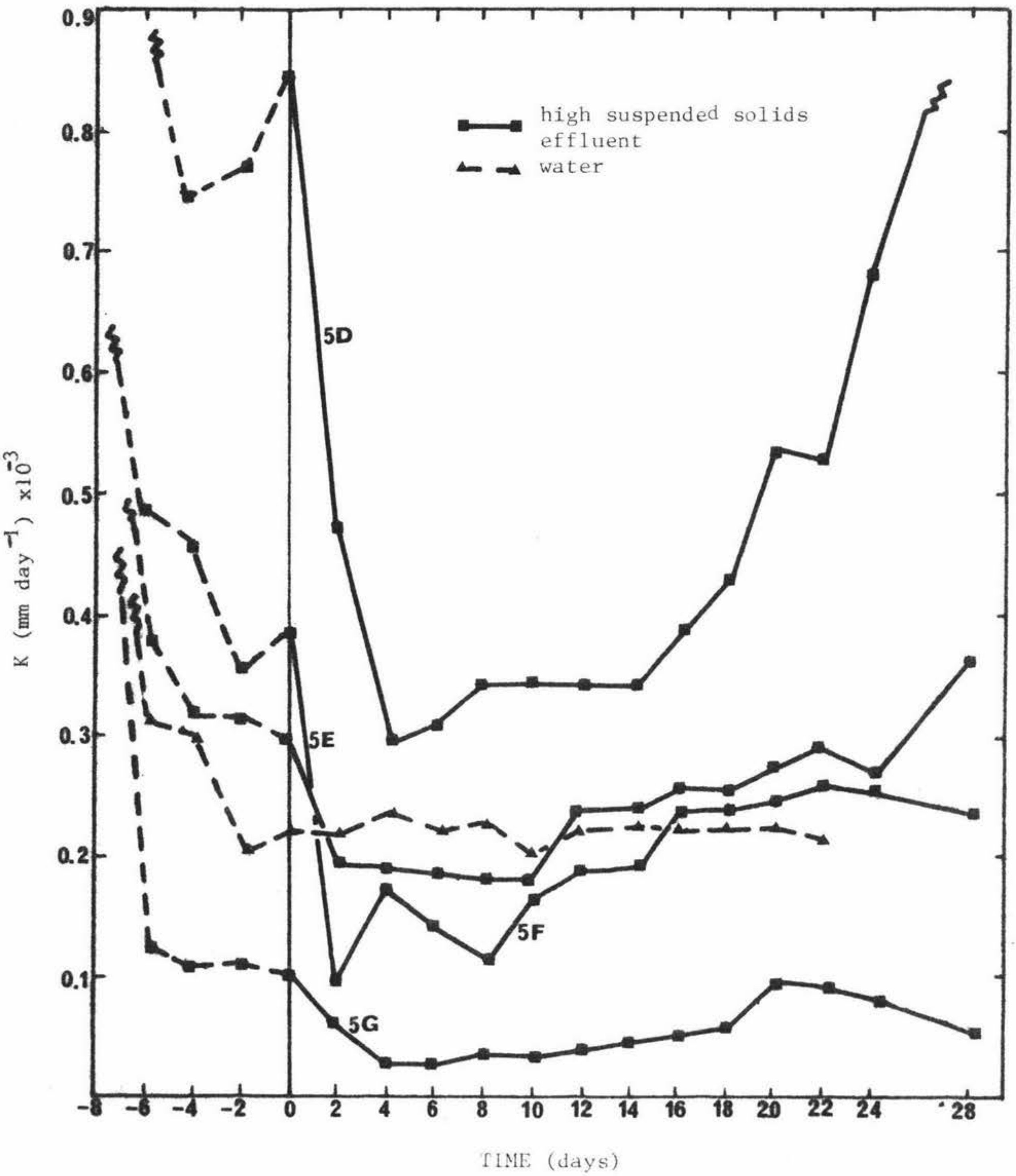

Fig. 5.28 Changes of K with time following application of high suspended solids effluent to four Tokomaru silt loam (100-200mm) cores, with a water only treatment core for comparison.

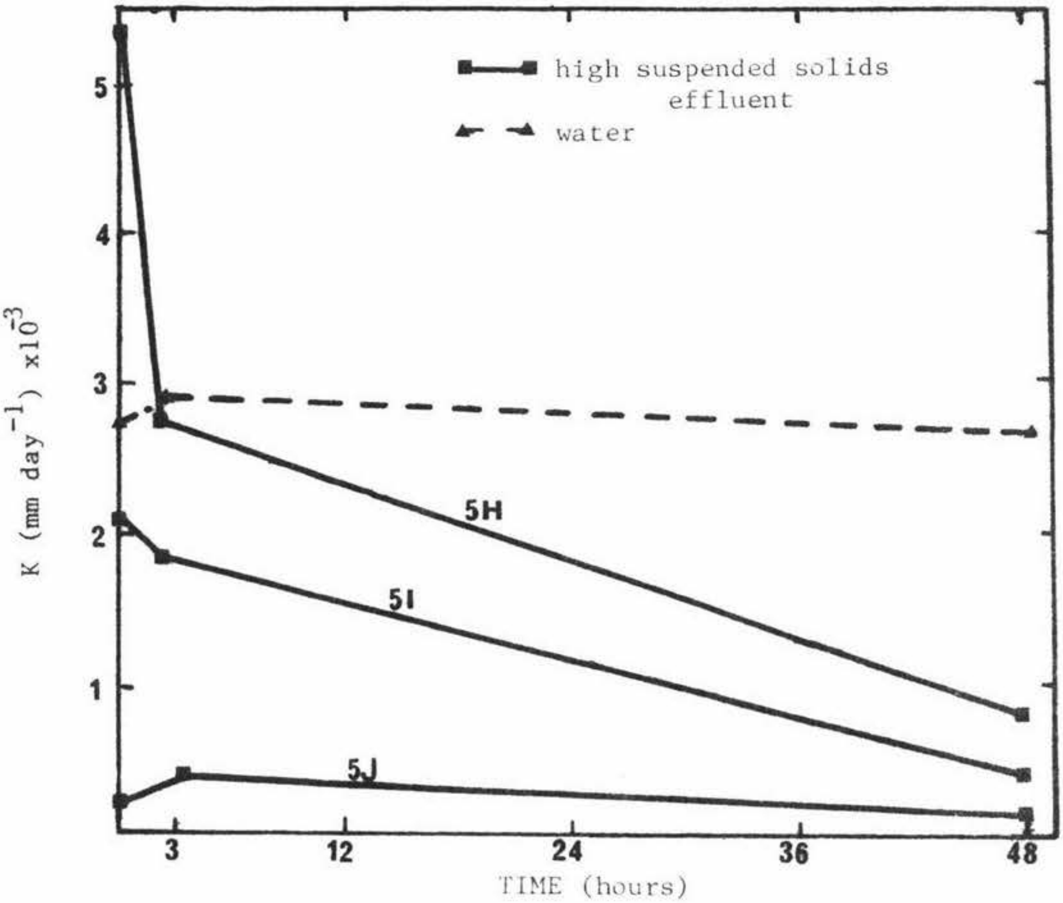

Fig. 5.29 Changes in K during ponding and over the two day period following an application of high suspended solids effluent to two Tokomaru silt loam (0-100mm) cores, with a water only treatment core for comparison.

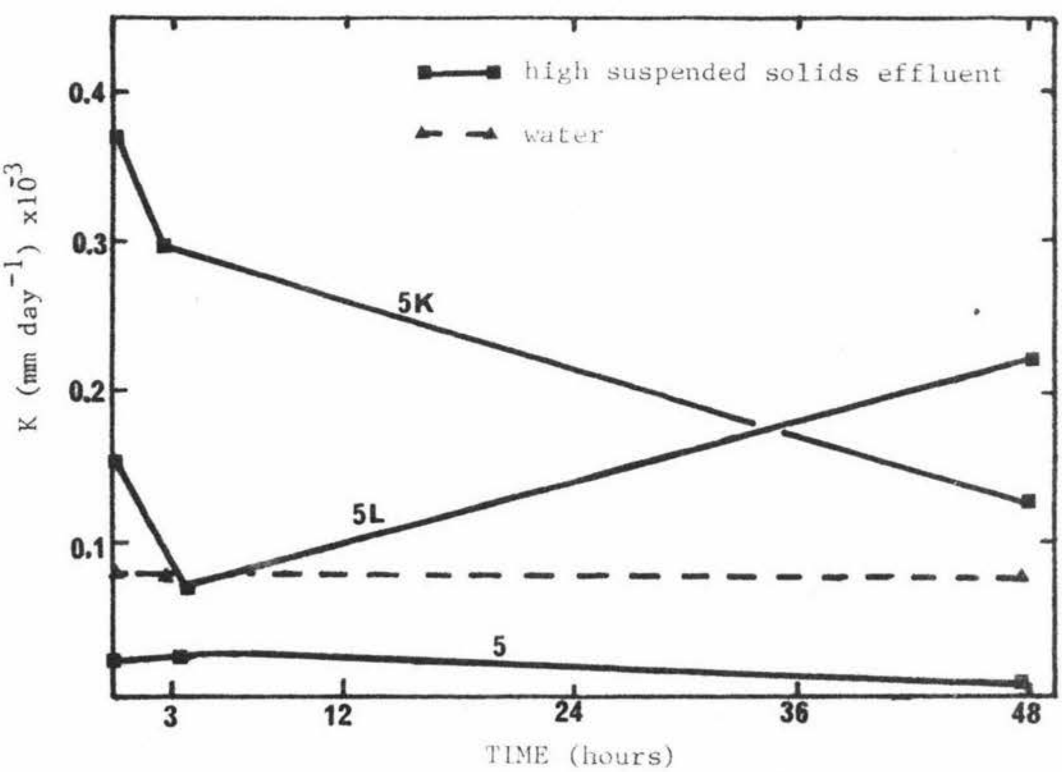

Fig. 5.30 Changes in K during ponding and over the two day period following an application of high suspended solids effluent to two Karapoti brown sandy loams (0-100mm) cores, with a water only treatment core for comparison.

two day period following the effluent application. As for the simulated whey treatment results (Fig. 5.22 - 5.25) this suggests that both physical and biological mechanisms were involved in the K decline.

The blockage with the high suspended solids effluent appeared to be more resistant to breakdown than for the simulated effluent. Whereas most cores receiving the simulated effluent exhibited virtually complete recovery within fourteen days, cores receiving the high suspended solids effluent took up to twenty-six days before recovering to a similar degree (Fig. 5.28). This difference in recovery rate between the two types of effluent probably reflects the relative decomposition rate of the suspended and dissolved solids by micro-organisms. Complete degradation of particulate matter would be expected to take longer.

5.4.6 A study of the increase in saturated hydraulic conductivity of "undisturbed" cores

The results obtained for K changes following effluent application to soil cores indicate that soil cores may exhibit K recovery; some cores even showed K values far greater than could be explained by removal of the effluent blockage alone. To evaluate the major mechanisms involved in the K increases a detailed study of individual core responses was carried out.

The formation of new conducting channels appeared to be confined to cores taken from the 0-100mm depth. Of the twenty-three subsurface cores studied, none showed an increase in K to above the "equilibrium K" value. In contrast, sixteen of the forty-five surface cores studied did exhibit such a response. It would appear, therefore, that the potential for recovery mechanisms to operate is higher for the surface layer cores.

The incidence of new flow channel formation appeared to be slower in cores receiving the microbial inhibitor. Of the fourteen surface cores receiving sodium azide only two exhibited a significant increase in K compared with fifteen of the thirty-two cores not treated with the inhibitor. This result, which demonstrates the importance of soil biological activity in K recovery, helps explain why the incidence of K increase was greatest in the surface layer cores (since surface cores would be more likely to contain channel-forming organisms).

Of the channel-forming organisms in soil cores, earthworms would be the group most likely to induce large increases in K. For this reason, it was decided to record both the presence of casting and the number of earthworms found in the cores at the completion of a conductivity experiment. This revealed that increases in soil K were closely related to the presence of earthworms. Fifteen out of the eighteen cores showing a large K increase were also found to contain live earthworms. In contrast, of the fifty-two cores not exhibiting a significant increase in K over the study period only three were found to contain live earthworms. It would appear therefore that earthworm activity was a major cause of K increase in the "undisturbed" core studies.

5.4.7 Use of the "double staining technique" to study blockage and recovery of channels in "undisturbed" cores

Results from core studies show that both large increases and decreases in K can occur over a relatively short period, indicating the dynamic nature of the soil pore system. To investigate rapid changes in conducting channels a double staining technique was employed using Methylene Blue and Rhodamine-B , as described in Section 3.1.6.

Methylene Blue was applied to cores after an eight to ten-day equilibration period. A further four to eight days was then allowed before the Rhodamine-B was applied. Following this second dye application, cores were sectioned and allowed to air dry before photographing. Figures5.31 - 5.35 are selected examples illustrating both the hydraulic flow pattern in "undisturbed" cores and the dynamic nature of the soil pore system. It is clear from Figs. 5.34 and 5.35 that

FIG. 5.31 A Section through a Tokomaru silt loam (100-200mm) core following staining with Methylene Blue (approximately to scale)

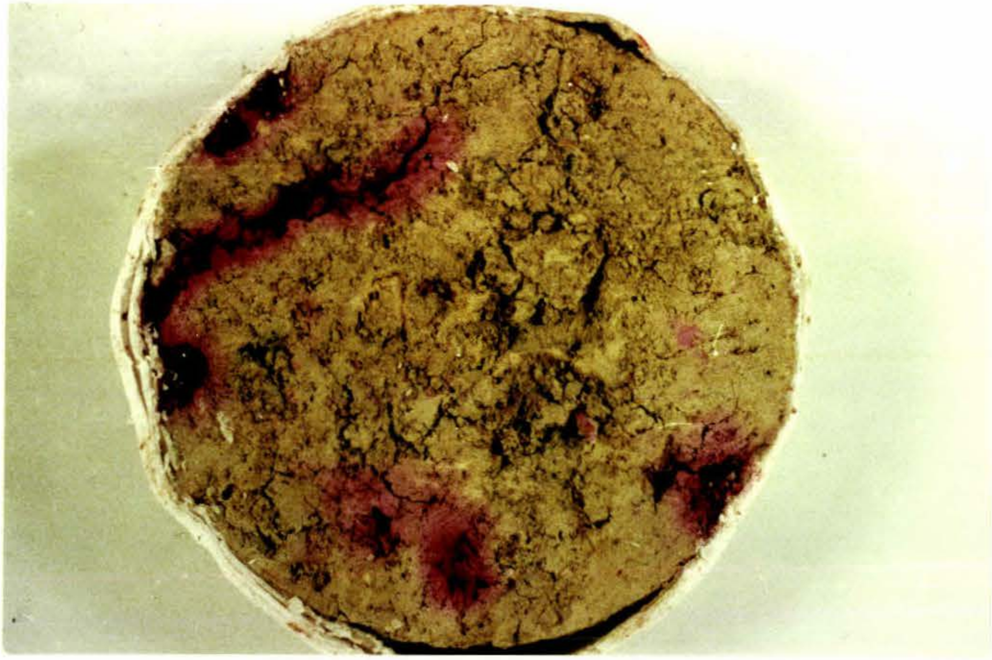

Fig. 5.32 A Section through a Tokomaru silt loam (0-100mm) core following staining with Rhodamine-B

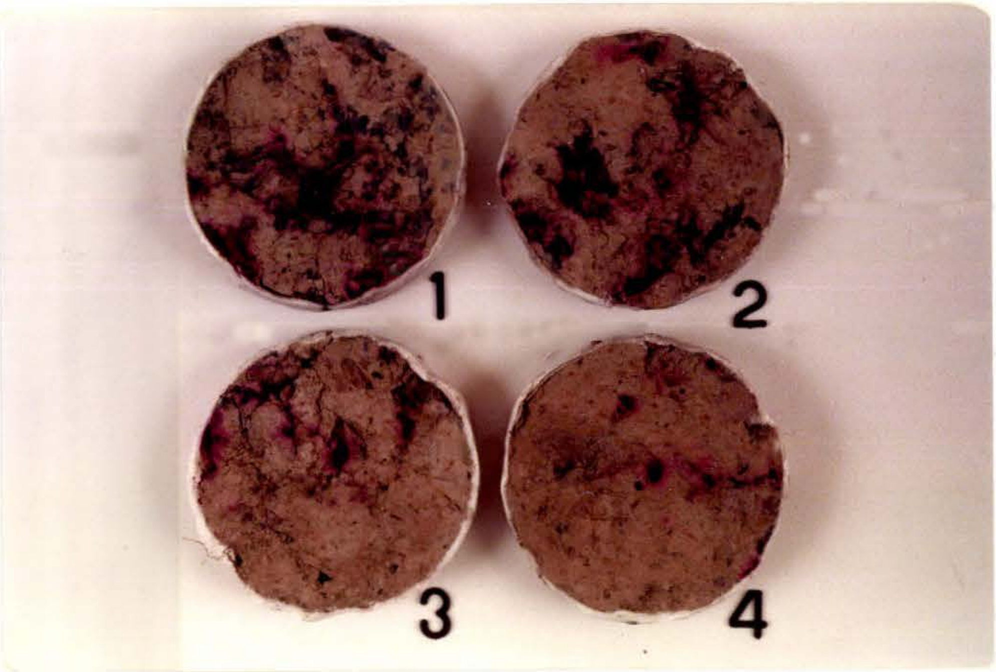

Fig. 5.33 Successive sections through a Tokomaru silt loam (0-100mm) core following "double staining", with layers numbered from the surface downwards.

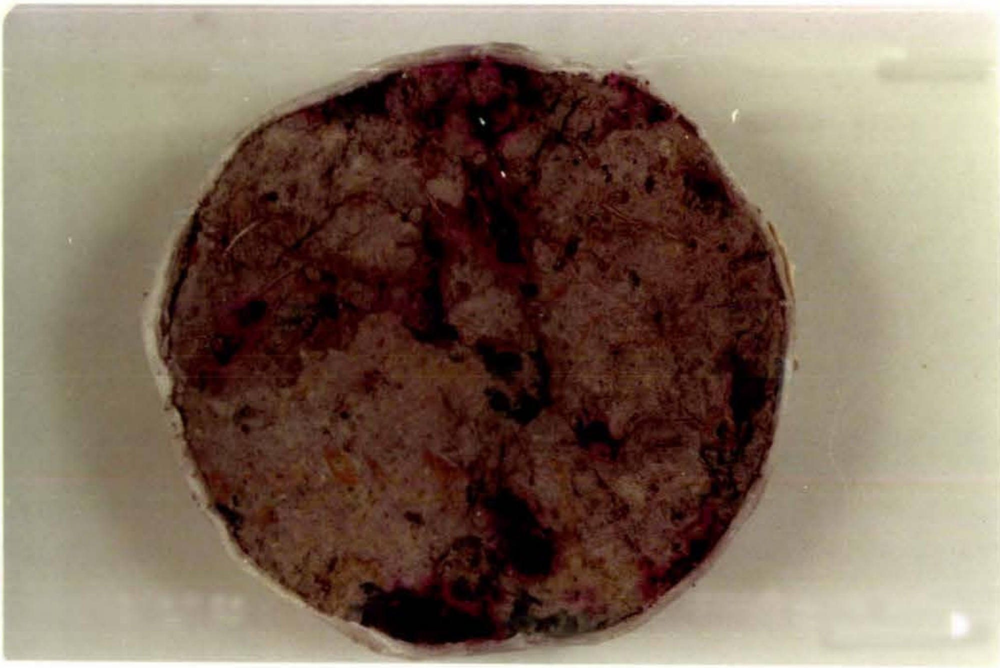

Fig. 5.34 A Section through a Manawatu silt loam (0-100mm) following "double staining".

Fig. 5.35   An enlarged Section of a Manawatu silt loam (0-100mm) core following "double staining".

a marked change in conducting channels has occurred in the interval between dye applications. It is likely that biological activity was the major cause of this change, since the most intense staining occurred in the regions where macrochannels (particularly earthworm channels) were in greatest abundance.

5.4.8 Information obtained from field studies

Using the double ring infiltrometer method (Section 3.1.5) an attempt was made to observe any changes in the water intake rate (termed the infiltration rate) of soils following spray irrigation of whey effluent to pasture. Three disposal sites, Te Rehunga, Longburn and Tokomaru were examined, with at least four recordings used for each treatment. For the control treatment, values obtained at the site prior to irrigation or values recorded at an adjacent non-irrigated site, were used. Infiltration values obtained at forty-five minutes were used for comparison between treatments since it was assumed that a "steady state" infiltration rate had been reached by this time.

The field study conducted at the Manawatu site was designed to detect any short term decline in infiltration rate following waste irrigation. Infiltration measurements were made at this site prior to, and two days after, an effluent application. The results obtained are shown in Table 5.2. Although a 30% difference was found between the

Table 5.2 Forty-five minute infiltration rates recorded prior to and two days after spray irrigation at the Longburn disposal site

Treatment	$\bar{x}$ (mm day^{-1})	s (mm day^{-1})	n	
Prior to irrigation	1600	713	6	n.s.
Two days after irrigation	1130	592	6	

mean values of the two treatments, the large variability in K meant that the results were not significantly different.

Field studies at the Te Rehunga and Tokomaru disposal sites were aimed at detecting long-term changes in the infiltration rate. At Te Rehunga, infiltration measurements were made during the middle of the spray irrigation season (November) and at the end of the spraying season (June 1st). The results obtained are presented in Table 5.3.

Table 5.3 Mid-season and end of season infiltration rates for the Te Rehunga disposal and control sites

Site	$\bar{x}$	s	n	
	(mm day^{-1})			
Mid-season				
Disposal	3970	1230	4	n.s.
Control	4100	1310	4	
End of season				
Disposal	4500	1200	4	n.s.
Control	4650	1460	4	

It is evident from the results that infiltration rates were similar at both the disposal and control sites and had not been reduced over the latter half of the spray irrigation season. This observation may be related to reduced hydraulic loadings during the part of the season resulting from a declining milk intake by the dairy factory.

The infiltration measurements at the Tokomaru site were made on an area of land which had recently been purchased by the dairy company and which had never before received wastes. Recordings were made prior to the initial waste application (June) and during the peak period of waste application (mid-September). The results obtained for these two recording periods are shown in Table 5.4. It is seen from the results that a very

Table 5.4 Pre-season and mid-season infiltration rates at the recently established Tokomaru disposal site

Time	$\bar{x}$	s	n	
	(mm day^{-1})			
Prior to irrigation	2150	560	4	***
Mid-season	198	48	4	

marked drop in the infiltration rate occurred within three months at this disposal site. The possible reasons for this decline are discussed in Section 7.2.3. This dramatic decrease in the infiltration rate over a relatively short time period emphasises the need for close integration of stock management with disposal practices, if site management problems are to be minimised.

5.4.9 Relevance of the "undisturbed" core studies to the field situation

Since the laboratory environment will not be identical to the *in situ* environment, results obtained from the "undisturbed" core studies will be of limited significance to the field situation, particularly results obtained for the rates and extent of blockage and recovery. The environmental parameters, particularly temperature and soil water content, may vary markedly throughout the year in the field, thus making it extremely difficult to design a laboratory environment accurately representing the field situation. Since the rates and extent of blockage and recovery depends on the environment (particularly the biological aspects) results obtained from field experiments may only be completely relevant to that site at a particular point in time.

However some conductivity studies using "undisturbed" cores can provide results of relevance to the field situation. Results obtained for different soils may enable a soil "ranking index" to be derived based upon the relative susceptibility of soils to blockage; for example, of the soil cores used in this study the most susceptible to blockage appeared to be the sub-surface Tokomaru silt loam cores. Also, information obtained on the mechanisms of blockage and recovery in "undisturbed" cores would, for the most part, be relevant to in situ studies. A problem exists however when interpreting the data on observed increases in soil K. Formation of a new flow channel in laboratory cores can cause a sudden large increase in K, particularly if a continuous channel is formed. Field measurement of K would not be expected to exhibit this sudden large increase, since the formation of a continuous channel through the soil profile is unlikely. Exceptions to this could occur however, such as when a new continuous pore is formed through a low permeability horizon, thus facilitating drainage.

5.5 Summary of Results

Whey effluent application caused a marked saturated hydraulic conductivity (K) decrease in "undisturbed" cores. A single 35mm equivalent application of simulated whey effluent caused an average change in K of -46% within the space of two days, compared to a mean change of less than -5% for cores receiving water only. Further applications of effluent invariably induced further reduction in K. Some cores receiving the repetitive effluent application exhibited almost complete blockage, with K changes exceeding -99%. However, total blockage or zero K was never attained.

Different soils exhibited differing responses to effluent application. This difference appeared to be partly related to the K values of cores, although both high and low K cores were found to be susceptible to blockage.

Recovery of K following blockage was observed. In cores receiving a single application of simulated whey effluent, followed by water every second day, virtual complete recovery of the initial K value("equilibrium K") was observed within two weeks. This suggests that a spray return interval of at least fourteen days is required in the field. The high suspended solids effluent appeared to induce a longer-lasting blockage, as a period of up to twenty-six days was needed before the "equilibrium K" was restored. This result illustrates the importance of maintaining low suspended solids levels in the wastewater. Some cores receiving repetitive applications of simulated effluent showed no sign of recovery within twenty-eight days of the final application.

Both biological and physical blockage mechanisms were identified in the blockage process, and both were observed to induce significant K decline. K increases in the soil cores appeared to be the result of both blockage removal from the pore space and creation of new conducting channels. Of the mechanisms likely to form new conducting channels, earthworm activity appeared to be the most important.

Although no significant data was obtained from the field studies, conductivity trends following effluent application were similar to the laboratory core observations. Long-term infiltration studies at one established disposal site indicated that this decline may only be temporary, however, since no noticeable decline in infiltration was observed over the spraying season. This result reflects the rapid recovery rate of soils and hence the dynamic nature of transmission properties. Information from a recently-established disposal site, where an order of magnitude decrease in infiltration rate was found within three months, suggests that a marked and possibly long-term decline in infiltration rate can occur with effluent application under some conditions at certain disposal sites.

CHAPTER SIX

LONG TERM CHANGES IN SOIL PROPERTIES RESULTING FROM THE APPLICATION OF DAIRY FACTORY EFFLUENT

CHAPTER SIX

LONG TERM CHANGES IN SOIL PROPERTIES RESULTING FROM THE APPLICATION OF DAIRY FACTORY EFFLUENT

6.1 Introduction

6.1.1 Scope of the study

The effluent disposal site is often regarded as a sink where the primary objective is to dispose of the wastewater with the minimum of disturbance to site management and environmental quality (Loehr, 1972). Much interest has been shown in recent years, from both an agricultural and environmental viewpoint, in the fate of the added waste components. Agricultural interest has largely been concerned with the fertilizer effect of the wastewater and what agronomic benefit may be obtained from the waste discharge. From the environmental viewpoint, the effects of the waste on groundwater or surfacewater quality is of interest, particularly in regard to possible enhancement of eutrophication or as a health hazard.

The solid component of dairy factory wastewater is largely organic in nature. These organic waste additions may be temporarily stored in the soil profile or may be rapidly decomposed by soil micro-organisms. This organic matter decomposition results in the formation of a large number of metabolic products ranging from simple organic and inorganic molecules to complex polymers and it is the fate of these breakdown products which has aroused particular interest.

6.1.2 Plant nutrient effects

Waste application to a soil can add large amounts of plant available nutrients and whey effluent application does just this. McDowall and Thomas (1961) reported that each cubic metre of undiluted casein whey contains the equivalent of 1.3kg of potassium sulphate, 8kg of ammonium sulphate and 2.7kg of superphosphate. Christie (1970) calculated that the spray irrigation of wastewater from the Toko dairy factory (Taranaki) applied the fertilizer equivalent per annum of 14 000kg ha^{-1} superphosphate, 5 300kg ha^{-1} muriate of potash and 4 800kg ha^{-1} sulphate of ammonia. The quantity of nutrients which these values represent may be appreciated when it is considered that normal fertilizer applications are at a rate of a few hundred kg ha^{-1} $year^{-1}$.

With these high nutrient application rates, it would be expected that the level of plant-available nutrients in the soil is increased. Such increases have been noted at several dairy factory spray disposal sites in New Zealand. At the Toko dairy factory site in Taranaki a four-fold increase in the phosphate soil test value and a ten-fold increase in potassium levels resulted after three years of spray irrigation (Christie, 1970). Parkin and Marshall (1976) recorded a seventeen-fold increase in phosphate and an eight-fold increase in potassium levels after five years of spray irrigation by a New Zealand dairy factory which manufactured casein. Wells and Whitton (1966) reported a five-fold increase in the plant-available phosphate and an eight-fold increase in the plant-available potassium levels after three years of irrigation with wastewater from a casein factory. Little research has been undertaken to examine the direct effects of dairy factory wastes on crop production, possible because actual disposal of the waste has been the primary concern, with relatively little attention paid to the agronomic benefits to be gained. Parkin and Marshall (1976) indirectly determined the increased herbage production by referring to the milk fat production figures for the disposal farm. They found that milk fat production at one disposal farm increased from 20 000kg $year^{-1}$ to 35 000kg $year^{-1}$ during five years of spray irrigation. However, no comparative control site data were given. Research conducted by Sharratt et al., (1959) examining the possible use of whey as a fertilizer source revealed that whey significantly increased the production of a grass crop during the second and successive seasons; the lack of response during the first season was attributed to the initially slow breakdown of nitrogen components in the whey. Wells and Whitton (1966) observed that pasture growth was stimulated by dairy factory waste application. This response was attributed to the high levels of nitrogen and potassium which were applied. Parkin and Marshall (1976) also noted from observation that dairy factory wastes stimulated plant growth. However, no experimentation was carried out by either Wells and Whitton or Parkin and Marshall to quantify these observations.

The quantity of macronutrients applied to the soil at a waste disposal site is usually far greater than the quantity removed from the soil by plant uptake (Pound and Crites, 1973). It is apparent, therefore, that either large losses of these nutrients occur to the groundwater and atmosphere, or that a substantial accumulation in the soil results, or both.

6.1.3 Effects on groundwater quality

Little research has been done to examine groundwater contamination beneath dairy factory disposal sites. This situation has prompted the Soil Science Department at Massey University, in conjunction with the New Zealand Dairy Research Institute, to examine groundwater quality at several disposal sites; this research is being undertaken at the present time. It is possible to obtain an indication as to the likely effect of dairy factory waste irrigation on groundwater quality by referring to results from other types of effluents. Most of this research has been based on nitrogen studies, since the nitrate form of nitrogen is both extremely mobile in soils (Thomas, 1970) and has evoked particular concern both as a hazard to health and as a cause of eutrophication (Stewart, 1970). A high proportion of the total nitrogen applied to the site may be leached to the groundwater. Adriano et al., (1975) for example, showed that over 70% of the total nitrogen applied to a site could be leached to the groundwater. Nitrate levels in groundwater below a disposal site may exceed the recommended public health levels of 10mg l^{-1} nitrate-nitrogen

(Pound and Crites, 1973).

Other workers, however, have found that there has been little detectable leaching of nitrate to the groundwater following wastewater application (Loehr, 1972; Murrman and Iskander, 1977). Thus it is evident that a wide variation exists between disposal sites in regard to potential groundwater contamination from nitrate. Factors such as the amount of nitrogen applied to and removed from the soil, the forms of nitrogen in the soil, the flux of water movement, and depth to the groundwater, are partly responsible for this variable response.

Compared with nitrogen, losses to the groundwater of other nutrients will be relatively minor. Phosphate, for example, is considered to be a relatively immobile nutrient since it is strongly sorbed to the soil fraction (Holford, 1974). Research shows that phosphate leaching losses are generally low (Adriano et al. 1975; Thomas, 1970). However, phosphate along with nitrate, is considered to be a key nutrient in eutrophication, therefore losses from the soil are important (Larson and Gilley, 1976).

6.4.1 Gaseous losses to the atmosphere

Losses of carbon, nitrogen and sulphur, from the soil to the atmosphere also occur. Again, little work appears to have been done to measure these losses from waste disposal sites largely because of the difficulty involved in the quantitative and qualitative analysis of the gaseous losses. Recording the rate of carbon dioxide production or oxygen consumption prior to, and following, waste application may however provide an indication as to the rate of organic matter breakdown as well as the fraction of organic waste which is lost to the atmosphere.

Addition of a readily oxidisable organic substrate to the soil, such as in whey effluent, would be expected to result in a marked increase in soil respiration rate. Work by Shields and Paul (1973) showed that 35% of carbon added as glucose to a soil was respired as carbon dioxide gas within two weeks. Since the solid component of whey effluent is primarily lactose (McDowall and Thomas, 1961), it would thus be expected that a large proportion of the added organic carbon would be decomposed within a few days of the application.

Reports of substantial gaseous nitrogen losses to the atmosphere have been cited following application of either waste or fertilizer nitrogen to soils. These losses involve both volatilization, which is the formation of ammonia gas by a chemical process from soil ammonium, and denitrification, which is a reduction process whereby nitrate is converted to gaseous nitrogen compounds. Carrie et al. (1958) found that gaseous losses accounted for 85% of the fertilizer nitrogen applied to a soil. Viets (1974) estimated the total gaseous nitrogen losses from a cattle feedlot to be approximately 50% of the nitrogen input. Several researchers believe that denitrification may readily occur at a liquid waste disposal site (Golberg, 1970; Pound and Crites, 1973). It is considered that waste application not only results in frequent additions of nitrogen and readily oxidisable carbon as a substrate for denitrifying bacteria, but the alternate wetting and drying cycles induced by the spray irrigation may provide ideal conditions for denitrification losses.

6.1.5 The nutrient balance

Nutrient balances for the soil profile at a waste disposal site show that nutrient inputs may greatly exceed losses (Pound and Crites, 1973; Murrman and Iskander, 1977). It is evident from such examples that either the components in the balance have not been determined with sufficient accuracy, or that the soil profile is acting as a store or sink for the applied nutrients. If the latter alternative is correct, it would be expected that a change in soil nutrient levels would be observed in a relatively short time period. Marked increases in the plant available forms of nitrogen, phosphorus and potassium have already been mentioned, but these would tend to account for only a small fraction of the applied nutrients. It is apparent therefore, that changes in the levels of nutrients in the organic fraction must occur for there to be significant accumulation.

Some examples of soil organic level increases following waste application have been cited. Uiga et al., (1977) found that soil organic carbon, nitrogen and phosphorus had been significantly increased as a result of municipal wastewater application to a disposal site in California. Up to a two and a half-fold change was found in total carbon levels down to a depth of 800mm. Murrman and Iskander (1977) also reported significant increases in soil organic carbon and nitrogen levels as a result of domestic wastewater application. Adriano et al., (1975) found that long term spray irrigation of food processing wastes had significantly increased total carbon and nitrogen in the surface 300mm and below 600mm depth; from 300-600mm a decrease in levels was observed. Although no explanation was given for this observation it may be that preferential denitrification occurred at this depth. Adriano et al., also noted that the carbon to nitrogen (C:N) ratio of soils was generally decreased following wastewater application. At one site, which had received milk powder factory wastes for ten years, they observed that the C:N ratios at various depths within the site profile had decreased from around 10:1 to as low as 3:1.

Research by Burke (1973) on soils of the Canterbury Plains, indicated that organic matter increases following waste water application may not be entirely due to the nutrients applied in the wastewater, but may also result from the irrigation effect of the wastewater. He estimated that twenty years of water only irrigation had increased the organic level in the soil profile by 13 000kg ha^{-1}. Conflicting results were obtained by Tate (1973) however who found that fifteen years of water only irrigation on a Canterbury Plain's soil resulted in a total carbon decrease from 4.4% to 3.7%. Tate found that a neighbouring soil which had received meatworks effluent for fifteen years showed a total carbon increase from 3.7% to 4.2%.

6.1.6 Soil pH

In early dairy factory spray irrigation systems, soil pH values were of interest as it was considered that the application to soil of acidic wastes (particularly lactic casein whey) would lower the pH (Eldridge, 1947). In contrast, a number of researchers have found that the long term spray irrigation of acidic dairy wastes actually induces a soil pH increase (McKee, 1955; Christie, 1970; Parkin and Marshall, 1976). The extent of the pH increase is determined by the rate and period of application, as well as the buffering capacity of the soil (Scott, 1962). Increases of over a pH unit have been observed (Sharratt et al., 1959)

A detailed study of the pH change following whey application to soil was made by Sharratt et al., (1959). In this study, it was found that whey induced an initial pH decrease which lasted for approximately twelve hours, after which a steady increase in pH was observed. This pH increase was considered to be due to the freeing of calcium and ammonium ions from the applied organic material.

6.1.7 Soil physical property changes

Research into bulk density and related physical property changes has largely been coupled with stock compaction studies; these are discussed in greater detail in Section 7.2.3. The increased hydraulic loading imposed on the soil by waste irrigation is generally observed to enhance any compaction effects which may occur, thus resulting in a higher bulk density (Pound and Crites, 1973; Pearson et al., 1975). However, some evidence exists which suggests that soil structure may be improved by waste application (Pillsbury, 1945; Scott, 1962). Sharratt et al., (1959) found that whey addition to a soil increased the degree of structural aggregation.

6.1.8 Soil biological property changes

The importance of earthworms is well recognised in New Zealand agriculture. In addition they are often considered to be the key to the efficient functioning of a disposal site (Yeates and Stout, 1975). In a paper reviewing research on earthworm abundance, Barley (1961) suggested that earthworms were advantageous not only from the soil fertility viewpoint but were also important modifiers of the soil physical properties, particularly from the drainage viewpoint. It would therefore be expected that a high level of earthworm activity at a disposal site would be an advantage.

No information appears to have been published on the effect of dairy factory effluent on earthworm numbers. However, Yeates (1976) and Yeates and Stout (1975) have reported on significant increases in earthworm numbers following dairy shed effluent application. The greater earthworm population at the disposal site was attributed primarily to the higher soil moisture content over the summer period reducing earthworm mortality. Availability of food supply and oxygen status have also been considered as determinants of earthworm numbers (Waters, 1955).

Since soil microbial activity is the prime factor in the degradation of organic waste material, several workers have undertaken research to examine the changes in microbial numbers and species composition following waste application to soil. It is evident from these studies that waste disposal fields do not possess a distinct microflora. Gilde et al., (1971) observed that cannery waste application resulted in an overall increase in microbial numbers. The species exhibiting increases were not solely confined to those capable of metabolising the waste components, but also included species which could utilise metabolites of the first set of microorganisms. They noted that microbial numbers were found to vary from field to field depending on food supply, temperature and soil types.

Di Menna (1966) found that dairy waste irrigation increased the number of yeasts in the soil by twenty fold. This response was found to be the result of the lactose in wastewater, since an effluent minus lactose treatment did not induce any change in numbers. Tewes and Gavel (1953) and Yeates and Stout (1975) have observed increases in soil bacteria numbers following application of dairy factory and dairy shed wastes. It was suggested by Doran et al., (1977) that the increased availability of oxidisable carbon, coupled with the reduction in soil oxygen status, were the two most important factors in determining changes in soil microflora following waste application. Ruscoe (1972) observed a distinct change in microflora with water only irrigation, thus suggesting that species composition changes with effluent application may be due in part to the irrigation effect.

6.1.9 Conclusion

In conclusion therefore it is apparent that an understanding of the fate of the waste material applied to a disposal site would be of considerable benefit for both agricultural and environmental purposes. From the agronomic viewpoint it would be advantageous to know what changes had occurred in soil properties as a result of waste application. Of the soil property changes which have been noted following dairy factory effluent application, most appear to be advantageous from the site management viewpoint. Perhaps of greatest importance to site management is crop production. Although many workers have observed large increases in the plant-available nutrient levels of disposal site soils, little work has been done to actually measure changes in pasture production. There also appears to be a general lack of information available on soil property changes at disposal sites. Because of this fact, farmers at present are often reluctant to allow wastes to be irrigated on their land. Consequently many factories seeking to install spray disposal systems are faced with the major problem of acquiring their own land.

From the environmental viewpoint most attention has been focused on possible groundwater contamination as a result of the waste application, particularly contamination with nitrate. Since groundwater contamination results from the soil's inability to retain or recycle all the applied nutrients, the long term functioning of sites is of interest, as the soil's ability to store applied nutrients would be expected to decline with time.

6.2 Experimental Design

As this section is primarily concerned with long term soil property changes resulting from waste application most of the results discussed are from established disposal sites at Te Rehunga and Tokomaru, both of which are alleged to have received wastes since 1963.

The soil parameters examined include bulk density, soil horizon depth, pH, inorganic nitrogen and phosphate levels, mineralisable nitrogen, total nitrogen and carbon, earthworm numbers and soil respiration. The analytical techniques used in the determinations are described in Chapter Two.

For a number of the soil properties examined it was important to record the time of sampling (both on a seasonal basis and in relation to the spray irrigation cycle) since a marked change in the level of some soil parameters can occur with time. This would be particularly applicable to plant-available nutrient levels and soil respiration rates.

6.3 Results and Discussion

In most cases results are presented in tabular form, as described in Section 5.4. For the chemical soil properties and soil respiration levels it is possible to use a "one-tailed t test" for statistical analysis since wastewater application is likely to induce undirectional changes in the levels of these parameters. For the remaining properties, however, observations suggest that either an increase or a decrease in the level of the parameter may be induced by waste application, hence a "two-tailed t test" was used.

6.3.1 Bulk density

Bulk density determinations were made on the surface horizons of the disposal and control sites at Te Rehunga and Tokomaru during the March-April period. The results of the analyses are shown in Table 6.1.

Table 6.1 Bulk density of the surface 100mm of soil from two disposal sites and corresponding control sites

Site	Treatment	$\bar{x}$	s	n	
		(kg m^{-3})			
Te Rehunga	Disposal	735	112	10	***
	Control	950	58	11	
Tokomaru	Disposal	1070	120	8	n.s.
	Control	1065	79	9	

The results show that a significant bulk density decrease has occurred in the surface layer of the Te Rehunga disposal site, while no change has occurred at the Tokomaru site. Both sites have been used as disposal farms for approximately the same length of time, hence these results illustrate the point that results obtained for one disposal site may not necessarily apply at another site.

The decrease in bulk density at Te Rehunga contrasts with most previous observations made on bulk density changes at wastewater disposal sites. More usually there is compaction due to soil wetness. Possible causes of the lower bulk density at Te Rehunga may be the increased organic matter content of the soil (Section 6.3.8), and the "pasture pulling" effect (Section 7.4). Pasture pulling could influence bulk density both directly by deposition of organic material at the soil surface and indirectly by stimulating soil biological and microbiological activity.

The low bulk density value at Te Rehunga may have a number of site management implications. Firstly, water storage capacity in the surface layer would be high since an inverse relationship exists between bulk density and porosity. A soil with a high surface storage would tend to have less surface runoff following high hydraulic loading rates. Secondly, soil infiltration characteristics would most likely be improved with a lowering of the bulk density (Hillel, 1971). Thirdly, a low bulk density indicates a lower bearing strength, hence pugging damage during wet periods may be more pronounced. This was found to be the case at the Te Rehunga disposal site (Section 7.2.3). Finally the bulk density decrease is likely to improve the aeration status of the soil profile.

6.3.2 Depth of the surface horizon

From observations made of the disposal and control site profiles at Te Rehunga, it was evident that a marked difference existed in the depths of the Ap horizons (a description of this horizon is presented in Table A1.1). Results of measurements taken to confirm this observation are shown in Table 6.2. The results show that the depth of the Ap horizon at the disposal site is 120mm on average, which represents a two-fold increase from the control site depth. An increase in the surface horizon depth would be expected to have occurred in conjunction with the bulk density decrease. However, when the results are expressed on a mass basis (computed as the product of density and soil volume) it is found that the depth of the Ap horizon is significantly greater than can be accounted for by bulk density change alone. It is apparent, therefore, that an appreciable organic matter accumulation has occurred hence a significant increase in the organic matter level would be expected.

Table 6.2 The depth of the Ap horizon at the Te Rehunga disposal and control sites

Treatment	$\bar{x}$ (mm)	s	n	
Disposal	120	15	10	***
Control	60	17	9	

6.3.3 Soil pH

pH in the surface 200mm was determined for the Te Rehunga disposal and control sites. Samples were collected approximately 24 hours after an effluent application and nearing the end of the spray irrigation season (May). The results obtained are shown in Table 6.3.

Table 6.3 Soil pH values for the Te Rehunga disposal and control sites

Depth	Treatment	$\bar{x}$	s	n	
0-100mm					
	Disposal	6.4	0.2	6	n.s.
	Control	6.4	0.25	6	
100-200mm					
	Disposal	6.8	0.14	7	***
	Control	6.0	0.18	7	
200-300mm					
	Disposal	6.7	0.19	7	***
	Control	6.2	0.15	7	

The results show that the pH of the disposal site soil at Te Rehunga was increased as a result of waste application, hence supporting the pH shift observed by other workers. No significant increase in pH was noted in the surface 100mm however. Since samples were taken soon after an effluent application, this result may suggest that surface soil pH values are initially decreased following whey effluent addition, but increase with time, as observed by Sharratt et al., (1959).

6.3.4 Soil nitrate and ammonium

Soil nitrate and ammonium levels were determined for the disposal and control sites at Te Rehunga, Tokomaru and Longburn. The inorganic nitrogen analyses were made to a depth of 300mm in order to observe the fertilizer value of the effluent. Samples were collected for analysis in June, 1978, approximately three weeks after the final effluent application for the season. The results obtained for each of the three sites are presented in Tables 6.4, 6.5 and 6.6.

From the results it is evident that the disposal site has markedly higher inorganic nitrogen levels than the control site soil, particularly the nitrate component. Soil nitrate levels were found to be increased by over ten fold at the Te Rehunga and Tokomaru sites. Even the Longburn site, which had been in operation for less than one season, showed a significant two-fold increase in the inorganic nitrogen level.

Since inorganic nitrogen analyses were made at only one point in time, seasonal changes in levels are not known, hence care must be taken in interpreting this data. The large differences in nitrate levels between the disposal and control sites do indicate however that nitrogen fertilizer supplements would not be required at the disposal site, even during the first season of operation.

Research conducted by the Soil Science department at Massey University has found that nitrate level differences between the control and disposal sites exist in the sub-surface soil layers as well. These observations indicate that nitrate may be leached to the groundwater at both the Te Rehunga and Tokomaru disposal sites (Pers. Comm. K.D. Earl, A.N. Macgregor).

Table 6.4 Nitrate and ammonium levels at the Te Rehunga disposal and control sites

Depth and Treatment	Nitrate $\bar{x}$	Nitrate s	Nitrate n	Ammonium $\bar{x}$	Ammonium s	Ammonium n
	($\mu gN\ g^{-1}$ dry soil)			($\mu gN\ g^{-1}$ dry soil)		
0-100mm						
Disposal	196	31	4 ***	30	10	4 ***
Control	46	1	4	8	1	4
100-200mm						
Disposal	24	3	3 ***	14	0.6	3 ***
Control	5	5	3	8	0.6	3
200-300mm						
Disposal	18	4	3 ***	12	0.5	3 ***
Control	5	2	3	7	0.4	3

Table 6.5 Nitrate and ammonium levels at the Tokomaru disposal and control sites

Depth and Treatment	Nitrate $\bar{x}$	Nitrate s	Nitrate n	Ammonium $\bar{x}$	Ammonium s	Ammonium n
	($\mu gN\ g^{-1}$ dry soil)			($\mu gN\ g^{-1}$ dry soil)		
0-100mm						
Disposal	83	4.5	3 ***	15	15	4 n.s.
Control	7	3.3	3	10	1.1	5
100-200mm						
Disposal	93	11	3 ***	10	0.6	3 n.s.
Control	6	1.1	4	10	0.5	4
200-300mm						
Disposal	79	8.5	4 ***	11	3.7	4 n.s.
Control	6	2.0	4	9	0.5	4

Table 6.6 Nitrate and ammonium levels at the Longburn disposal and control sites

Depth and Treatment	Nitrate $\bar{x}$	Nitrate s	Nitrate n		Ammonium $\bar{x}$	Ammonium s	Ammonium n	
	(μgN g^{-1} dry soil)				(μgN g^{-1} dry soil)			
0-100mm								
Disposal	42	14	4	***	14	0.9	4	**
Control	18	2.3	4		9	2.4	4	

6.3.5 Plant-available phosphate

"Plant-available" phosphate levels were determined for surface horizons of the Te Rehunga and Tokomaru disposal and control sites. Sampling was undertaken at the end of the spray irrigation season (June). The results of these analyses are shown in Table 6.7.

Table 6.7 "Plant-available" phosphate levels for the disposal and control sites at Te Rehunga and Tokomaru

Site	Depth	Treatment	$\bar{x}$	s	n	
			(μg P g^{-1} dry soil)			
Te Rehunga	0-100mm	Disposal	660 9	128	6	***
		Control	76	13	6	
	100-200mm	Disposal	320 12	82	4	***
		Control	24	6	4	
Tokomaru	0-100mm	Disposal	252 5	29	5	***
		Control	55	13	5	
	100-200mm	Disposal	278 9	23	4	***
		Control	30	7	4	
	200-300mm	Disposal	151 7	42	5	***
		Control	19	3	4	

As for the soil nitrate, the inorganic phosphate levels were markedly higher for the disposal site than for the control site, thus indicating that inorganic phosphate had accumulated as a result of waste application. The Olsen values obtained for the two disposal sites are extraordinarily high since values of 20-30μg of phosphate per gram of soil are normally cited for agricultural soils. Even the control site soils at both disposal sites have relatively high inorganic phosphate levels. Since workers have found herbage at disposal sites to be nutrient enriched (Wells and Whitton, 1966), and because the disposal and control sites were situated on the same farm, it is likely that a transfer of fertility has occurred from the disposal site via the stock.

6.3.6 Potentially mineralisable nitrogen

The mineralisable nitrogen analyses were made on surface soil samples from all three disposal sites and their corresponding control sites. The results obtained are shown in Table 6.8.

Table 6.8 Potentially mineralisable nitrogen values for the disposal and control sites at Te Rehunga, Tokomaru and Longburn

Site	Treatment	$\bar{x}$	s	n	
		(μgN g^{-1} dry soil)			
Te Rehunga	Disposal	716	298	4	***
	Control	170	30	4	
Tokomaru	Disposal	172	45	12	**
	Control	144	26	12	
Longburn	Disposal	216	26	8	n.s.
	Control	224	35	8	

From Table 6.8 it is seen that both the Te Rehunga and Tokomaru disposal sites exhibit higher potentially mineralisable nitrogen than both the corresponding control sites, thus indicating that mineralisable nitrogen levels are increased with long term waste application. The difference at Te Rehunga is of a greater magnitude, however, with the mineralisable nitrogen value at this disposal site being four times the control site level.

Mineralisable nitrogen increases indicate that either there has been a change in the microbial availability of the total organic nitrogen, or a significant increase in the total nitrogen pool, or both. Total nitrogen results from Te Rehunga (Table 6.9) indicate that both alternatives may operate for that site. No significant differences in mineralisable nitrogen are observed at the Longburn site due to the short history of wastewater application.

6.3.7 Total nitrogen

Total nitrogen levels for the disposal and control sites at Te Rehunga and Tokomaru were determined to a depth of 600mm. The results of these determinations are presented in Tables 6.9 and 6.10.

Table 6.9 Total nitrogen levels at the Te Rehunga disposal and control sites

Depth	Treatment	$\bar{x}$ (% dry soil)	s	n	
0-100mm	Disposal	1.15	0.17	9	***
	Control	0.55	0.08	6	
100-200mm	Disposal	0.72	0.09	3	***
	Control	0.33	0.02	3	
200-300mm	Disposal	0.57	0.01	3	***
	Control	0.27	0.02	3	
300-400mm	Disposal	0.42	0.02	3	***
	Control	0.13	0.02	3	
400-500mm	Disposal	0.25	0.04	3	***
	Control	0.10	0.01	3	
500-600mm	Disposal	0.23	0.02	3	***
	Control	0.10	0.01	3	

Table 6.10 Total nitrogen levels at the Tokomaru disposal and control sites

Depth	Treatment	$\bar{x}$	s	n	
		(% dry soil)			
0-100mm	Disposal	0.41	0.04	9	n.s.
	Control	0.41	0.04	10	
100-200mm	Disposal	0.32	0.02	10	***
	Control	0.27	0.04	10	
200-300mm	Disposal	0.27	0.02	8	**
	Control	0.17	0.04	9	
300-400mm	Disposal	0.12	0.01	9	**
	Control	0.10	0.01	9	
400-500mm	Disposal	0.08	0.01	10	**
	Control	0.09	0.01	10	
500-600mm	Disposal	0.07	0.01	10	n.s.
	Control	0.07	0.01	10	

The results from both sites show that waste-treated soil has significantly higher levels of total nitrogen thus suggesting that waste application has enhanced nitrogen accumulation. The Te Rehunga site again exhibited the most pronounced difference with all depths of the profile showing at least a two-fold increase in total nitrogen from the control site values. The greater total nitrogen level down to 600mm in this disposal site profile is computed as being equivalent to an increase of 16 500 kg ha^{-1}. Based on wastewater analyses, it is estimated that approximately 45 000kg of nitrogen ha^{-1} has been applied to the Te Rehunga disposal site over the fifteen years of operation (Pers. Comm. A.N. Macgregor). Thus it appears as though approximately one-third of the nitrogen applied to the site may have been retained in the soil profile.

Significant differences in the sub-surface total nitrogen levels suggest that nitrogen has moved down the profile. For Te Rehunga, total nitrogen differences between the control and disposal site occur to a depth of

600mm. No samples were taken from below this depth. Increases in total nitrogen at the Tokomaru site were observed to a depth of 400mm only. In contrast, samples taken from the 400-500mm depth at the Tokomaru disposal site exhibited a lower total nitrogen level than the control site samples. Although only conjecture, it is possible that a total nitrogen decrease occurred at this depth due to enhanced denitrification. The increased hydraulic loading at the site plus continual additions of a readily oxidisable carbon supply (data in Section 6.3.8 indicates that carbon is leached to this depth) may provide an ideal environment for biological denitrification to occur.

6.3.8 Organic matter

Organic matter levels were estimated from calculated total carbon data, as described in Section 3.2.1. The results obtained for estimated organic matter levels at the Te Rehunga and Tokomaru disposal and control sites are presented in Tables 6.11 and 6.12.

Results show that organic matter levels were significantly higher at both the Te Rehunga and Tokomaru disposal sites than for the corresponding control sites. The organic matter difference at the Te Rehunga site represents a total of 247 000kg ha^{-1} of organic matter to 600mm depth. It appears therefore that a large quantity of organic matter may have accumulated in the soil profile as a result of waste application. This accumulation has occurred over the range of soil depths examined with the greatest absolute gain being observed in the 200-300mm depth.

At Tokomaru, the organic matter difference in the surface 600mm between the control and disposal site is computed as being 31 000kg ha^{-1} of organic matter which represents only one eighth of the difference observed for Te Rehunga. Comparing the two disposal sites, it appears that either there has been a marked difference in the amount of waste material applied to each, or that the Tokomaru soil has a greater potential to recycle the

Table 6.11 Organic matter levels at the Te Rehunga disposal and control sites

Depth	Treatment	$\bar{x}$	s	n	
		(% dry soil)			
0-100mm	Disposal	19.74	1.70	12	***
	Control	14.36	1.60	9	
100-200mm	Disposal	13.30	2.10	4	***
	Control	7.60	1.10	4	
200-300mm	Disposal	10.54	1.65	3	***
	Control	4.66	0.90	3	
300-400mm	Disposal	6.31	0.72	5	***
	Control	2.58	0.43	4	
400-500mm	Disposal	4.86	1.02	5	***
	Control	1.84	0.17	5	
500-600mm	Disposal	4.00	0.44	5	***
	Control	1.25	0.15	5	

Table 6.12 Organic matter levels at the Tokomaru disposal and control sites

Depth	Treatment	$\bar{x}$	s	n	
		(% dry soil)			
0-100mm	Irrigated	8.92	2.10	19	n.s.
	Control	8.23	0.84	19	
100-200mm	Irrigated	6.10	0.53	10	***
	Control	4.99	0.45	10	
200-300mm	Irrigated	3.45	0.46	10	***
	Control	2.37	0.41	10	
300-400mm	Irrigated	1.92	0.25	10	**
	Control	1.66	0.14	10	
400-500mm	Irrigated	1.22	0.19	10	n.s.
	Control	1.25	0.07	10	
500-600mm	Irrigated	0.98	0.08	10	n.s.
	Control	1.00	0.11	9	

applied organic wastes, or a combination of both.

No significant organic matter increase was observed in the 0-100mm depth at the Tokomaru site, suggesting that a steady-state organic matter level (where input equals output) may have been reached in this soil layer. Since waste application did not appear to significantly change the organic matter level in the surface layer, it is apparent that the rate of nutrient cycling in this layer had been increased.

It is also apparent that carbonaceous compounds are leached through the soil profile since significant organic matter accumulation was detected to at least a depth of 600mm at Te Rehunga and 400mm at Tokomaru. Laboratory studies employing "undisturbed" soil cores were used to verify this leaching phenomenon. The results of these studies (Table A4.1) show that carbohydrate material was readily leached through soil cores, thus indicating that significant quantities of readily oxidisable carbohydrate may move through the soil profile.

An examination of the carbon to nitrogen (C:N) ratio for Te Rehunga shows that for most soil depths the C:N ratio is smaller for the disposal site soil than for the control. This observed change indicates that the nitrogen fraction of the applied effluent accumulated at a faster rate than the carbon fraction of the waste.

6.3.9 Earthworm numbers

Worm numbers were determined for all three disposal sites and their corresponding control sites. Sampling was carried out in early June when it was expected that nearly all earthworms would be near the soil surface (Yeates, 1976). It was therefore assumed that the numbers recorded in the surface 100mm in June represented the total number of earthworms in the soil profile.

Table 6.13 shows the worm count results from the three sites.

Table 6.13 Earthworm numbers in the surface 100mm of soil at the Te Rehunga, Tokomaru and Longburn disposal and control sites

Site	Treatment	x	s	n	
		(number m^{-3}) $x 10^{-3}$			
Te Rehunga	Disposal	9.8	5.5	20	***
	Control	1.6	2.4	20	
Tokomaru	Disposal	15.2	7.8	10	***
	Control	3.2	3.5	10	
Longburn	Disposal	18.1	8.0	20	n.s.
	Control	21.3	9.0	10	

The results show that the worm numbers at the Te Rehunga and Tokomaru disposal sites are five to six-fold higher than the corresponding control sites, thus illustrating that long-term waste irrigation has had the effect of increasing earthworm numbers. No change in numbers was apparent at the Longburn site, indicating that the build-up in numbers observed for the remaining two sites occurred progressively over time. In addition, it is noted that the Longburn control site exhibited a substantially greater number of earthworms than the other two control sites, thus illustrating that marked spatial variation in earthworm numbers can occur.

An examination of earthworms collected from Te Rehunga indicated that spray irrigation may also induce a change in species composition and age-group distribution. Earthworms from the species Allolobophora caliginosa and Lumbricus rubellis were evident at both the control and disposal sites, but earthworms from the species Octolasium cyaneum were found at the control site only. Whereas earthworms found at the control site were of a relatively similar age group, those found at the disposal site were of a variety of age groups. It appears, therefore, that the irrigation has altered the breeding pattern, possibly by permitting breeding to occur over an extended season.

6.3.10 Soil respiration

A preliminary respiration experiment was designed to determine whether or not a change in the soil respiration rate was induced by waste application to a soil. Samples were taken from the Te Rehunga control and disposal sites at approximately seven and fourteen days following an effluent application. The soil respiration values after three hours of incubation are shown in Table 6.14.

Table 6.14 Oxygen consumption rates in soil samples taken from the Te Rehunga disposal and control sites at seven and fourteen days after waste application

Time	Depth	Treatment	x	s	n	
			(moles $O_2 min^{-1} kg^{-1}$	dry	soil)$\times 10^{5}$	
7 days	0-100mm	Disposal	6.60	1.37	4	***
		Control	2.50	0.60	4	
	100-200mm	Disposal	6.60	0.28	2	
		Control	1.25	0.16	2	
	200-300mm	Disposal	1.67	0.75	2	
		Control	0.38	0.20	2	
14 days	0-100mm	Disposal	2.20	0.67	4	n.s.
		Control	2.50	0.60	4	
	100-200mm	Disposal	1.00	0.50	2	
		Control	1.67	0.16	2	

From the day seven oxygen consumption values, it is evident that microbial activity is stimulated by effluent application. It is likely that this stimulation effect is relatively short-lived however, as the disposal site oxygen consumption values on day fourteen are not significantly different from the control site values. This observation could be due to exhaustion of oxidisable substrate although the pcssibility of some other limiting factor cannot be discounted.

A second experiment was conducted to examine the rapidity and magnitude of microbial response to a whey effluent application. This experiment involved applying simulated whey effluent to soil samples and recording changes in oxygen consumption over time. Although only results obtained for the Te Rehunga site are shown here, data obtained for the Tokomaru and Longburn sites (Figs. A4.3 and A4.4) indicate a similar trend. Soil samples from both the disposal and control sites at Te Rehunga were used. The mean changes in oxygen uptake with time for the two replicates per treatment are shown in Figs. 6.1 and 6.2.

It is evident from the results that the soil respiration rate is markedly increased by the addition of a readily oxidisable carbon source. Statistical analysis of the difference in respiration rate between the whey effluent and water treatments shows that the results are significant to the 95% confidence level. Increased oxygen uptake in the whey effluent treatment samples occurred within a few hours of the application, thus indicating the rapidity of microbial response to effluent addition. The effluent application to soil samples from the disposal site induced a more rapid response than did application to the control site soil. This suggestes that a "pre-conditioning" effect may be evident whereby micro-organisms capable of metabolising the whey effluent components have been increased in numbers as a result of previous waste applications.

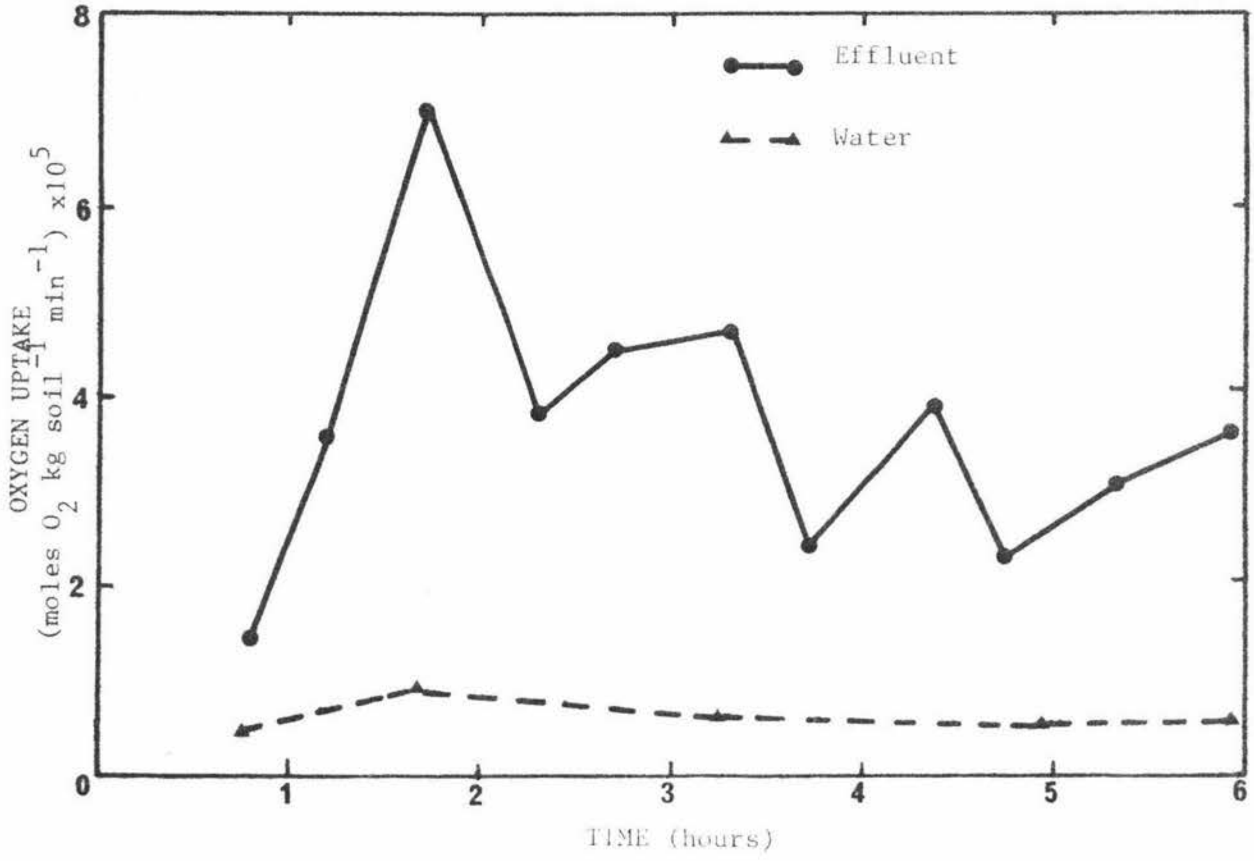

Fig. 6.1 Changes in soil oxygen consumption with time following effluent and water applications to soil samples from the Te Rehunga disposal site.

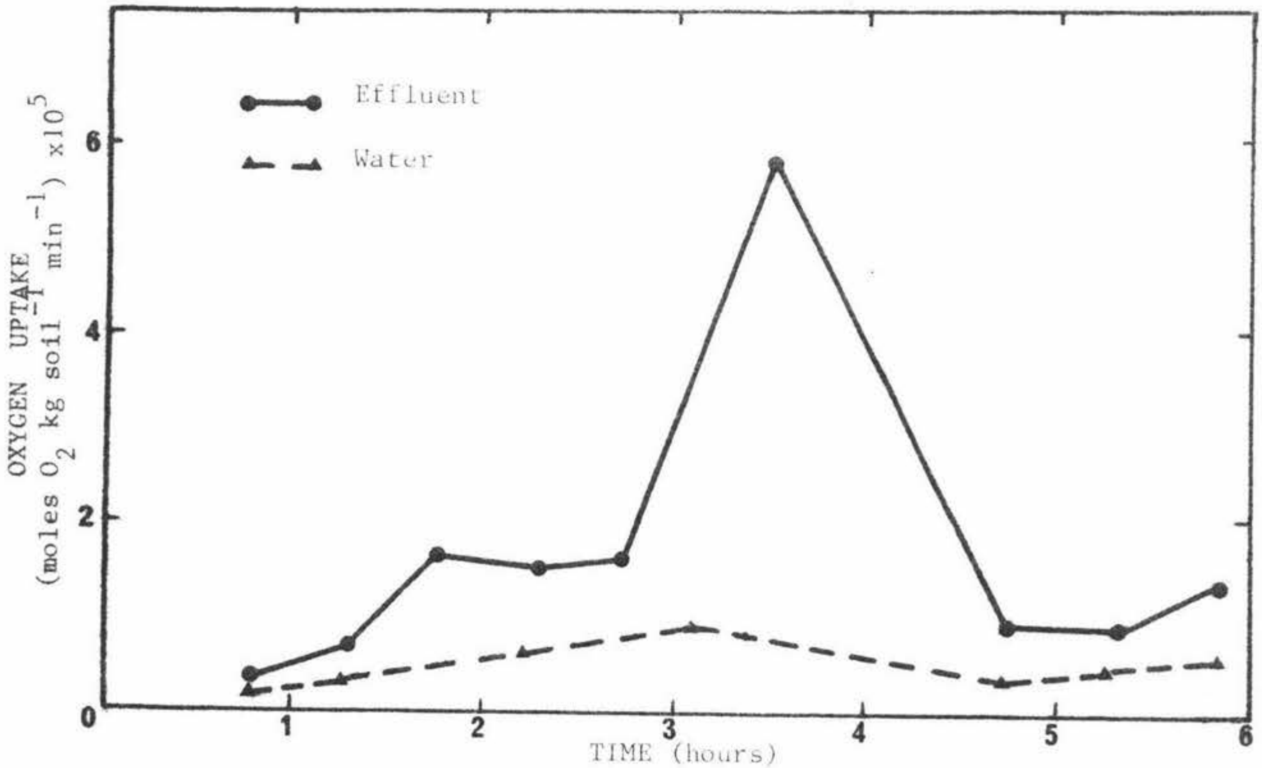

Fig. 6.2 Changes of soil oxygen consumption with time following effluent and water applications to soil samples from the Te Rehunga control site.

6.4 Summary of Results

Differences between the disposal and control site levels for a number of soil physical, chemical and biological parameters were found. The bulk density in the surface 100mm of the Te Rehunga disposal site was found to be lower than that of the control site ($735 kg\,m^{-3}$ compared with $950 kg\,m^{-3}$). No bulk density difference was evident at the Tokomaru site, however. The depth of the Ap horizon at the Te Rehunga disposal site was found to be twice the control site depth of 60mm.

The pH values for the 100-300mm depth at the Te Rehunga disposal site were significantly greater than for the control site with typical values for the sites being 6.7 and 6.0, respectively.

Nitrate levels were substantially higher at both the Te Rehunga and Tokomaru disposal sites than at the corresponding control sites, with up to ten-fold differences being recorded. Even the Longburn disposal site, which had received wastes for less than one season, had surface nitrate levels twice the control site level. Significantly higher ammonium levels were evident at the Te Rehunga and Longburn disposal sites than at the corresponding control sites, although these differences were much smaller than for nitrate.

Both the Te Rehunga and Tokomaru disposal sites exhibited large differences in the "plant-available" phosphate levels between the surface layers of the disposal and control sites. Over ten-fold differences in "plant-available" phosphate levels were found between the disposal and control sites, with a typical value for the disposal site being $300 \mu gP\ g^{-1}$ of oven dry soil.

Significant differences in potentially mineralisable nitrogen were found between the disposal and control sites at Te Rehunga and Tokomaru. This difference for the Te Rehunga site was much greater, however, with the disposal site level of $716 \mu gN\ g^{-1}$ dry soil four times that of the control site.

The mineralisable nitrogen differences between the disposal and control sites may be explained in part by total nitrogen differences, since total nitrogen levels were found to be higher at the disposal site. Significant differences in total nitrogen levels between the disposal and control sites were observed for all depths sampled at Te Rehunga (0-600mm) and for the 100-400mm depth at Tokomaru. The total nitrogen difference between the Te Rehunga disposal and control site represents an accumulation of around 16 000kg ha^{-1} for the fifteen years of waste application to this site.

Significant organic matter differences between the disposal and control sites were found for all depths sampled at Te Rehunga and for the 100-400mm depth at Tokomaru. For Te Rehunga this difference was approximately eight times that found at Tokomaru and represented an accumulation over the fifteen year irrigation period of 250 000kg ha^{-1}. Relating the total carbon data to the total nitrogen information it appears as though the waste application may have narrowed the carbon:nitrogen ratio for most soil depths at the Te Rehunga disposal site.

Earthworm studies revealed that numbers at the Te Rehunga and Tokomaru disposal sites were six-fold and approximately five-fold higher than numbers at the respective control sites. No difference in numbers was found at the Longburn site.

Soil respiration rates were found to be significantly higher at the Te Rehunga disposal site than at the control site seven days after effluent application. Fourteen days after the application however no significant difference between the two sites was apparent. Simulated whey effluent application to soil samples was found to induce a marked increase in the respiration rate. This response occurred within a few hours of the whey application and appeared to be quicker in samples from the disposal sites than those from the control site.

CHAPTER SEVEN

MANAGEMENT PROBLEMS AT DAIRY FACTORY DISPOSAL SITES

CHAPTER SEVEN

MANAGEMENT PROBLEMS AT DAIRY FACTORY DISPOSAL SITES

7.1 Introduction

From a farming viewpoint, the spraying of dairy factory wastes onto heavily-stocked pastoral land invariably induces management problems. The major problems to be considered include poor drainage, pasture burning, pasture pulling and stock diseases. For each of these topics information obtained from the Te Rehunga, Tokomaru and Longburn sites will be presented along with other cited case studies.

7.2 Soil Drainage

The hydraulic loading onto the disposal site is often considered to be the principle design factor in spray irrigation system design (Lawton et al. 1959; Pound and Crites, 1973; Parkin and Marshall, 1976). As mentioned in Section 4.1, excess soil water not only limits production from the site but also impedes site management. The improvement of site drainage characteristics will therefore increase management efficiency.

Before an attempt is made to improve soil drainage, it is necessary to determine the cause of the problem. Primarily the distinction must be made as to whether the problem is the result of a high groundwater table within pervious soil materials, a perched water table over an impermeable sub-surface layer, or low permeability through fine-textured soil horizons. It is then necessary to identify the causes of either the water table or low permeability problem.

7.2.1 Te Rehunga - A high groundwater table drainage problem

A high groundwater table problem is generally seasonal in nature with marked fluctuations occurring between periods of water surplus and water deficit. At dairy factory effluent disposal sites the period of greatest hydraulic loading onto the site generally coincides with the period of water surplus and a high water table (Figs. 4.1 and 4.2), thus compounding any drainage problem.

Surface ponding is observed at Te Rehunga over much of the July-October period (Pers. Comm J. Renwick). To determine the reason for this ponding the infiltration rate at the disposal site was measured in mid-June and early July. On both occasions, the mean infiltration rate was found to be high (over 3000mm per day). Studies using groundwater access tubes installed along an east-west transect at the disposal site indicated that the cause of ponding over the period was most likely to be a high groundwater table.The water table was found to rise to a level at or near the soil surface over the July-October period. The gravelly soil horizons (Bw1 and Bw2) appear to function as an unconfined aquifer for underground water movement, with the water originating in the Ruahine range situated to the west of the disposal farm.

The solution to the Te Rehunga site drainage problem is undoubtedly the installation of a suitable sub-surface drainage system in order to control the height of the water table. Although an interception drain was installed along the western boundary of the farm in 1972, this has not proved totally effective and further sub-surface drainage is required.

7.2.2 Longburn - An infiltration problem

As was mentioned in Section 5.1.2, several reports have been cited where spray irrigation schemes have failed when the application rate and frequency were based on the pre-irrigation soil infiltration rate. Infiltration measurements made in May at the Longburn disposal site indicated a high soil infiltration rate (with a mean value of over 3000mm per day), thus suggesting that an infiltration problem would be unlikely. After a few months of spray irrigation, however, a severe drainage problem was observed at this site. The disposal farm was slightly undulating which resulted in a number of shallow depression regions of approximately five to ten metres

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in diameter. It was in the depressions that ponding was observed during the spraying season (Fig. 7.1). Prior to waste irrigation ponding occurred in these hollows only after high intensity storms (Pers. Comm. P. Snoxell). Not only were these ponded depressions a hinderance to farm management, but they also created an odour problem which aroused the concern of neighbours.

In order to determine the cause and extent of the drainage problem in these hollows, infiltration studies were conducted. These studies showed a mean infiltration rate in the hollows of less than 20mm per day. Also, the soils in the depressions were found to be unsaturated below 50mm depth, thus it appeared as though the drainage problem was caused by sealing at or near the soil surface.

The most likely cause of this surface sealing was perhaps the high effluent loading applied to the depressions. Although the application rates measured were not excessive (Appendix A2.3), it was observed that a substantial volume of liquid flowed into these depressions as a result of "backflow" from the sprinkler pipeline when the pump was switched off. Another factor contributing towards surface sealing appeared to be casein fines and other suspended solids material in the waste liquid, since particulate matter from the wastewater was often observed to coat the soil surfaces of hollows (Fig. 7.2). Analysis of effluent samples from the Longburn factory (Table A5.1) showed that the wastewater contained up to 0.4% suspended solids.

To obtain a more comprehensive understanding of the surface sealing process at Longburn, laboratory studies were undertaken using "undisturbed" cores. The first experiment aimed to determine the extent of K reduction following application of the high suspended solids effluent to soil cores collected from the elevated areas of the Longburn site (Karapoti silt loam cores). The results from this experiment are presented in Section 5.4.5 and illustrate that the high suspended solids effluent can render cores almost completely impermeable.

Fig. 7.1 A ponded depression area at the Longburn disposal site

Fig. 7.2 Effluent solids material coating the soil surface following spray irrigation.

A second laboratory experiment was designed to examine the extent of the blockage in cores from the depression areas. Using the basic design procedure described in Section 5.3, the K values of six depression soil cores and four cores from the elevated area were recorded over time. The surface layersof all cores (this included both the top and base 10mm of the core) were removed on day 4 and again on day 8. All six cores from the depressions showed remarkably similar response in terms of K change, thus it was considered that a single mean value change could be used for result presentation. The mean value changes of the two groups of soil cores employed are shown in Fig. 7.3; a semi-log scale is used in order to present the data from both groups of cores on the same graph.

The results obtained from the depression area cores illustrate three distinguishing features. Firstly, the K value of these cores differ markedly to that of the cores from the elevated areas. This difference is both in terms of the initial K values (where a ten-fold difference was found between the mean values of both groups of cores) and in the response to removal of the core surface. Secondly, the removal of the core surfaces induced a marked increase in K, thus supporting previous observations that the blockage in hollows results from surface sealing. The third noticeable feature was the comprehensive decline in K following the K increase induced by core surface removal. This observation indicates that some blockage mechanisms operate following the exposure of a fresh core surface. It is hypothesised that perhaps this blockage was a biological process resulting from the stimulation of aerobic soil micro-organisms in the region of the freshly exposed surface.

In order to obtain evidence to support the above hypothesis oxygen uptake studies were conducted using soil cores from the depression areas. From these studies it was seen that very little aerobic respiration occurred in these cores. However, by breaking open the cores to expose fresh soil surfaces to an aerobic environment a greater rate of oxygen uptake was

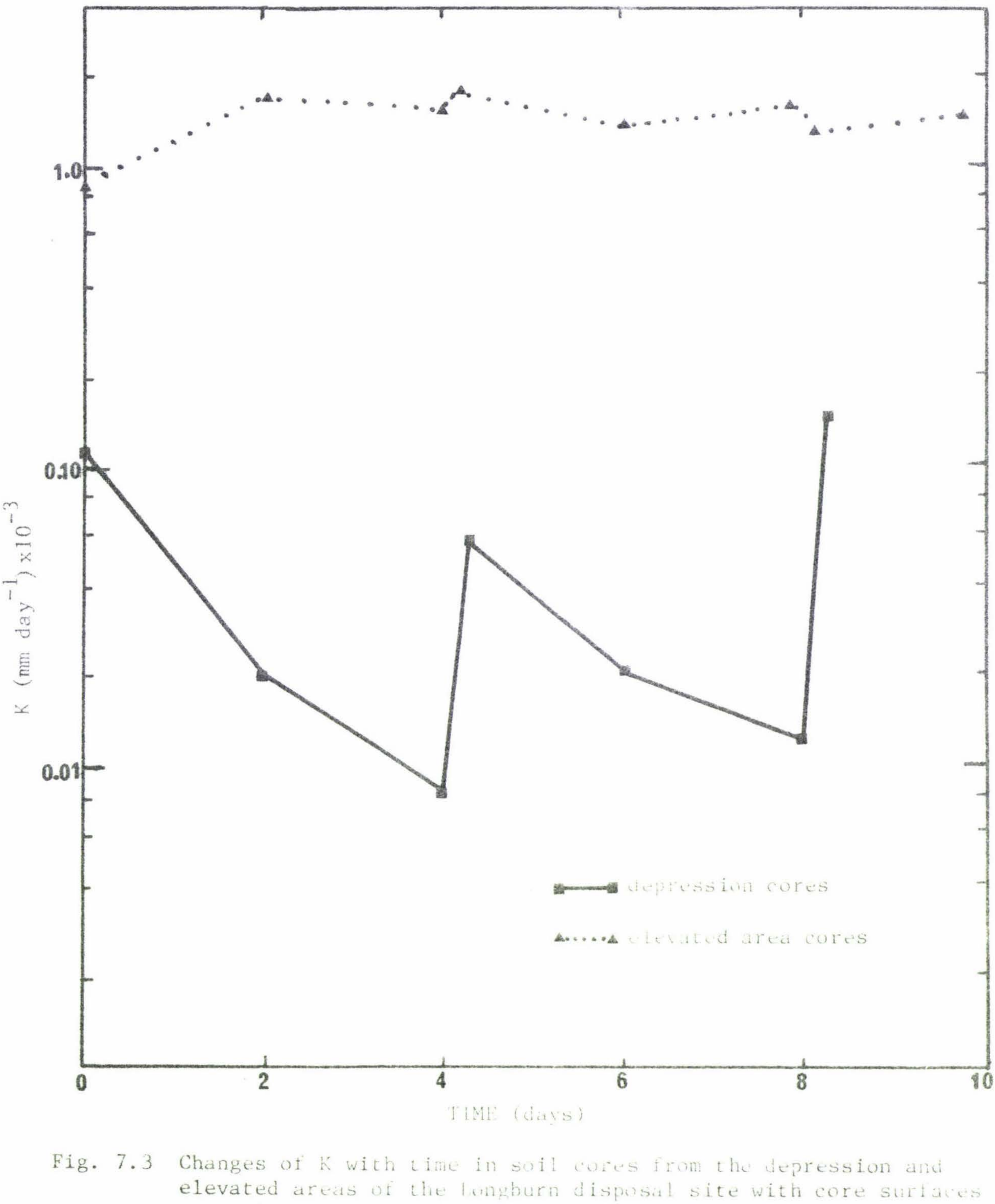

Fig. 7.3 Changes of K with time in soil cores from the depression and elevated areas of the Longburn disposal site with core surfaces removed on days 4 and 8.

observed. It was therefore evident that oxygen was limiting microbial activity in the depression soils. Removal of the core surface would thus have enabled microbial activity to occur at or near the exposed surface, with the corresponding metabolic products of this increased activity as the likely cause of channel blockage.

Further analysis of the depression soils at Longburn revealed an absence of earthworms. The death or migration of earthworms from these hollows was probably due to oxygen stress, although other factors cannot be discounted. Since earthworms appear to be of major importance in soils recovering from low hydraulic conductivity (Section 5.4.6), it is probable that the low infiltration rate in the hollows will be a relatively long term phenomenon.

7.2.3 Integration of stock management with the disposal system

The two factors most likely to cause a lowered infiltration rate at a disposal site are effluent blockage of conducting channels (discussed in Chapter 5), and pugging or treading damage of the soil surface (Pound and Crites, 1973). A number of researchers have examined the effect of animal traffic on soil physical properties, and have found that marked decreases in water and air permeability can occur as a result of pugging (Edmond, 1958; Tanner and Mamaril, 1959; Laycock and Conrad, 1967; Pearson et al., 1975).

Treading damage is invariably a greater problem in wet soils than in dry soils (Laycock and Conrad, 1967), thus any treading damage is likely to be compounded at a waste disposal site. This point was illustrated by Pearson et al., (1975) who found that set stocking a food processing waste disposal site with cattle resulted in an infiltration rate reduction from 500mm day^{-1} to 70mm day^{-1} over a six month period.

In order to determine the importance of stock management on soil infiltration, studies were undertaken on two adjacent fields at the Longburn disposal site; field 1 had been rotationally grazed prior to waste application, whereas field 2 had been lightly set stocked for approximately six weeks prior to the infiltration study. Both fields had last received waste approximately one week prior to the study. The results obtained are shown in Table 7.1.

Table 7.1 Infiltration rates at a rotationally grazed disposal site (field 1) and at a set stocked disposal site (field 2)

Site	$\bar{x}$	s	n	
	(mm day^{-1})			
Field 1	932	840	8	*
Field 2	445	480	8	

The results show a marginally significant difference in infiltration rate between the two fields. It is likely that treading damage under the set stocked management system was a major factor contributing to this difference. However, since infiltration rates at field 2 were not measured prior to set stocking, this cannot be stated conclusively.

A further indication of the effect of stock management on soil infiltration rates may be obtained by examining the data obtained from the newly established disposal site at Tokomaru (Table 5.4). These results show that the infiltration rate at the disposal site dropped from over 2000mm day^{-1} to less than 200mm day^{-1} in the space of three months. It is likely that pugging contributed significantly to this rapid decline since the area was set stocked for much of this three month period. Stock compaction may have been compounded at this site since the area was cultivated and resown prior to waste application.

From observation it would seem that set stocking may be an unwise management practice at a spray disposal site if pugging is likely to be a problem. It would appear that rotationally grazing the pasture immediately prior to spray irrigation is a more suitable management practice.

7.3 Pasture Burning

Pasture burning, whereby the cover crop becomes scorched and yellow in appearance, has been observed following application to the soil of dairy factory wastewater (United States Environmental Protection Agency, 1971; Parkin and Marshall, 1976). Pasture burning not only results in a decreased crop production, but can also encourage weed ingression in the scorched areas (McDowall and Thomas, 1961).

A number of factors have been implicated as causing pasture burning from dairy factory wastewater. Parkin and Marshall (1976) noted that the high sodium ion content in milk-powder factory wastewater may cause burning. They suggested that recommendations made by Wilcox (1949) should be adhered to in order to prevent burning. These recommendations were that the total concentration of cations in the wastewater should not exceed 25 milli-equivalents litre^{-1}, and the sodium ion content should not be more than 80% of the soluble cations present. Burning has been attributed to high pH of the wastewater resulting from high levels of caustic soda (Marshall and Parkin, 1976); caustic soda is used as a cleaning agent in the dairy industry. McDowall and Thomas (1961) reported that burning may occur from whey effluent which had been stored for several days before spray disposal. Finally, work by Scott Russell (1977) indicates that oxygen stress may also cause pasture burning. He found that plant injury through oxygen stress occurred following addition of organic material to the soil.

Pasture burning to at least some degree was observed at all three study disposal sites, although the Tokomaru site appears to have experienced the most severe burning problem. In several areas of this farm large patches existed where complete pasture death had occurred over the summer period

(Fig. 7.4). This burning was attributed to high caustic soda levels in the effluent, due to inadequate mixing of casein wash-water with the Factory wash water. Following this incident steps were taken by the Tokomaru dairy factory to ensure that any high pH material would be adequately diluted with the washdown water.

At Longburn, pasture burning was noted primarily in the depressions experiencing drainage problems. The pasture in many of these depressions had been completely burnt off and regrowth appeared to be slow. It is likely that oxygen stress following ponding may have been the major cause of this burning. Oxygen may have become limiting in a relatively short time period in the ponded depressions, since there would not only be a high oxygen consumption by soil micro-organisms utilising the effluent solids, but the rate of oxygen replenishment would be severly limited by ponding. Osmotic stress, as a result of excessive salt concentration, may also have been a factor in this pasture burning. Although electrical conductivity values from the Longburn dairy factory effluent samples (Table A5.1) indicate that the wastewater was unlikely to induce water stress, high evapotransporation rates in summer may have resulted in a high soil solution salt concentration.

Pasture burning has been observed at Te Rehunga (Pers. Comm. J. Renwick). It was noted that this burning was mainly over the summer period, particularly in windy conditions. This observation suggests that the burning is largely a salt effect, since the environmental conditions described indicate a high evapotranspiration rate, thus an increased salt concentration both on the foliage and in the root zone.

7.4 Pasture Pulling

Pasture pulling refers to the uprooting of pasture plants during stock grazing. It occurs when the tensile strength of the stems is greater than the root anchorage. A summary of the research conducted on pasture pulling is presented by Thomas and Evans (1975). They noted that the

Fig. 7.4 Pasture burning at the Tokomaru disposal site

problem generally occurred on free draining soil, particularly after heavy rain following a dry spell. The incidence of pasture pulling was reported as being higher in shallow-rooted plant species, particularly if they were growing in a highly fertile soil. It was also noted that pasture appeared to be more susceptible in late spring and autumn and over the summer period, when the plants were of a more stalky nature.

Of the three disposal sites examined only the Te Rehunga site had a pasture pulling problem (Fig. 7.5.). This appeared to be a major factor limiting herbage production at the site particularly over the spring period. Site studies reveal that three main factors may induce the pasture pulling. Firstly, plant root growth was largley confined to the surface horizons; very few roots were observed below 300mm. It is likely that the high soil fertility and constantly high moisture content are the main reasons for this shallow rooting habit since there would be no requirement for a deep rooting system. A further cause of the shallow rooting habit is likely to be the high ground water table over the winter-spring period (Section 7.2.1). Secondly, the low bulk density of the surface Ap horizon (Section 6.3.1) indicates that plant roots in this soil layer are likely to have poor anchorage. Thirdly, plants at the disposal site are observed to have a 'clumpy' growth habit (which may largely be due to stock pugging effects). Thomas and Evans (1975) note that clumps of grasses tend to be more prone to pasture pulling than do single tiller plants.

One means by which the pasture pulling at Te Rehunga may be alleviated is to encourage deeper rooting. This may possibly be achieved by spelling the disposal site over a relatively dry period, thus encouraging plant root extension as water is sought. Perhaps the most effective means of encouraging deeper rooting is to install a suitable drainage system to control the height of the water table, thereby reducing the incidence of root rot. A further means by which the pasture pulling damage may be reduced

Fig. 7.5 Pasture pulling at the Te Rehunga disposal site.

at least temporarily is to resow the disposal site pasture, thus changing the dominant growth habit from a clumpy to a single tiller form. Resowing may also allow deeper rooting grass species to be introduced. Since cultivation may be difficult at Te Rehunga, due to the presence of a stony soil horizon at approximately 150mm (Table A1.1), zero cultivation techniques may be necessary if resowing is considered.

7.5 Stock Health

It has been suggested by some workers that the spray disposal of dairy factory wastes may result in ill-thrift or stock disease. Sanborn (1953) considered that regular irrigation of dairy factory wastes may lead to a build-up of virulent tubercle bacilli in the soil, thus perpetuating tubercullosis if dairy cows are grazed on the area. Larson and Gilley, (1976) voiced concern about high nitrate levels in disposal site pasture. They noted that high nitrate levels in feed may be responsible for milk production decline, loss of appetite, abortion and poor stock thrift. Based on soil and herbage analysis at several New Zealand dairy factory disposal sites Wells and Whitton (1966) suggested that a supplementary phosphate or trace element supply may be needed to provide a better balanced diet for stock.

In spite of these concerns, however no real stock health problems appear to have been encountered in New Zealand as a result of spray irrigating dairy factory wastes (Parkin and Marshall, 1976). Cristie (1970) has observed facial eczema and hypomagnesaemia problems at two Taranaki dairy factory disposal sites, but these problems were alleviated by changing site management practices and by feeding magnesium supplements. Observations at several disposal site farms indicate that dairy factory effluent irrigation reduces the incidence of bloat (Pers.Comm. K.D. Earl).

No stock health problems have been encountered at either of the established disposal sites, Te Rehunga and Tokomaru. The farm manager at

Te Rehunga does however supply magnesium supplements to his stock as a precautionary measure.

At Longburn the main milking herd initially appeared reluctant to graze the disposal site pasture. This palatability problem was overcome in time however with no observable effect on stock thrift (Pers. Comm. P. Snoxell).

7.6 Summary

A number of management problems have been encountered at dairy factory disposal sites, and the Te Rehunga, Tokomaru and Longburn sites are not exceptions. The most notable of these problems are: poor site drainage, pasture burning and pasture pulling.

Drainage problems were observed at the Te Rehunga, Longburn and recently-established Tokomaru disposal sites, with the cause of the drainage problem at each site being different. The drainage problem at Te Rehunga resulted from a high groundwater table, thus indicating the need for an effective sub-surface drainage system. Surface infiltration problems were evident at the recently established Tokomaru site and in depression areas of the Longburn site. Observations and and measurement at these two sites indicate the infiltration rate decrease to result from both effluent blockage and pugging.

Pasture burning from dairy factory effluent appears to be a common occurrence at disposal sites since all three sites examined had experienced the problem. In comparison, pasture pulling appears to be a more localised phenomenon with the Te Rehunga site being the only cited case.

No stock health problems were encountered at the three disposal sites examined.

APPENDICES

APPENDIX I

PROFILE DESCRIPTIONS

Table A1.1 Profile description of the Kopua silt loam at the Te Rehunga disposal site

Horizon designation	Horizon depth (mm)	Colour	Texture	Structure
Ap	0 - 50	dark yellowish brown (10 YR3/4)	silt loam	fine crumb
Ap2	50 - 150	dark yellowish brown (10 YR3/4)	slightly stony silt loam	medium to fine nut
Bw	150 - 300	brown (10 YR4/3)	moderately stony silt loam	very fine nut
Bw2	300 - 400	yellowish brown (10 YR5/4)	slightly gravelly silt loam	very fine nut
C1	450 - 900	light olive brown (2.5Y5/3)	slightly gravelly silt loam	very fine blocky
C2	900 +			

Table A1.2 Profile description of the Karapoti brown sandy loam at the Longburn disposal site

Horizon designation	Horizon depth (mm)	Colour	Texture	Structure
Ap1	0 - 100	very dark greyish brown (10 YR3/2)	sandy loam	nut
Bw	200 - 300	olive brown (2.5Y5/4)	sandy loam	nut
C	300 +	olive (5Y5/4)	sand	

Table A1.3 Profile description of the Kairanga fine sandy loam at the Tokomaru disposal site

Horizon designation	Horizon depth (mm)	Colour	Texture	Structure
Ap	0 - 200	very dark greyish brown (2.5YR3/2)	silt loam	fine nut
Bwg	200 - 600	grey (5YR5/1) with yellowish brown (10YR5/4)	silt loam	fine to medium blocky
Cr	600 +	grey (5YR5/1) with yellowish brown (10YR5/6) staining	silt loam	medium prismatic and fine blocky

Table A1.4 Profile description of the Manawatu silt loam

Horizon designation	Horizon depth (mm)	Colour	Texture	Structure
Ap	0 - 250	Brown (10YR 4/3)	silt loam	nut
Bw	250 - 750	Light olive brown (2.57 5/4)	silty loam to fine sandy loam	nut and block
C	750 +	Olive brown (2.57 4/4)	fine sand	

APPENDIX II

BASIC DATA AND CALCULATIONS RELATING TO THE WATER BALANCE STUDIES

A2.1 Rainfall

The daily rainfall figures for Te Rehunga and the computed daily water balance data for Palmerston North were supplied by the New Zealand Meteorological Service. Because of the close proximity of Longburn to Palmerston North and similarity in elevation, it was assumed that the Palmerston North water balance could be applied to the Longburn disposal site. In order to obtain a comparison between the rainfall pattern at Te Rehunga, Tokomaru and Palmerston North, mean monthly rainfall figures for the period 1941-1970 were obtained from the New Zealand Meteorological Service (1973); these are given in Table A2.1. For the purposes of this study the rainfall pattern at Palmerston North and Tokomaru is sufficiently similar to warrant the usage of Palmerston North water balance data at the Tokomaru site.

A2.2 Evapotranspiration

The "potential evapotranspiration" values used in the water balance calculations are shown in Table A2.2. These values were determined by Coulter (1973) using Penman's method of evapotranspiration estimation. Since "potential evapotranspiration" refers to evapotranspiration from an extensive area of land covered by a short green crop which is never water stressed, it should provide a reasonable estimate of pasture evapotranspiration at spray disposal sites in a humid climate.

A2.3 Effluent Application

Because of the lack of information on the volume of effluent produced by the Te Rehunga and Longburn dairy factories, it was necessary to approximate the effluent component of the water balance by measuring application rates and estimating the period of application and the return interval.

Table A2.1 Mean monthly rainfall data (in mm) for Te Rehunga, Tokomaru and Palmerston North over the period 1941-1970

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Te Rehunga	109	99	107	112	132	152	137	132	102	140	124	150	1500
Tokomaru	97	76	84	84	99	112	102	97	79	102	91	119	1142
Palmerston North	84	69	74	74	86	99	91	84	69	89	79	104	1002

Table A2.2 Average estimated potential evapotranspiration values for pasture at Palmerston North (mm month^{-1})

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
128	103	80	47	24	14	18	31	53	82	106	123	809

Table A2.3 Calculation of the available water holding capacity and readily available water

(W is the mm of water held in the root zone)

	Te Rehunga	Longburn	Tokomaru
Rooting depth (mm)	300	600	600
W at -0.1 bar	157	267	-
W at -1.0 bar	114	-	-
W at -15 bar	86	144	-
Available water holding capacity (mm)	71	123	-
Readily available water (mm)	43	-	-

Effluent application rates were measured using plastic raingauges with a catchment area of $0.0036m^2$. Eight readings were made per site, with the gauges placed at varying distances from the sprinkler nozzles. The average rate of effluent application was found to be approximately 30mm $hour^{-1}$ at both the Longburn and Te Rehunga sites. From this application rate,plus information obtained on the length of application and the return interval, it was estimated that the total annual volume of effluent irrigated was in the region of 1000mm.

For the purposes of the water balance, a reasonable assumption to make is the volume of effluent produced by a factory is directly proportional to the quantity of milkfat processed (McDowall and Thomas, 1961). By referring to the seasonal dairy production curve in Fig. A2.1, approximations may then be made as to the volume of effluent produced per month.

A2.4 Soil Information

For both the Te Rehunga and Longburn water balances some information on the water retention properties of the soils was required. This information was obtained using the methods of analysis described in Sections 3.1.1 and 3.1.3, for determination of soil bulk density and soil water retentivity characteristics, respectively. The results obtained are shown in Figs. A2.2, A2.3, A2.4 and A2.5. Rooting depth estimates were based upon soil profile examinations.

From the water retentivity curve and rooting depth information, the "available water holding capacity" (AWC) could be derived using Equation 3.2. The "readily available water storage capacity" at the Te Rehunga site was also estimated. This involved replacing the permanent wilting point volumetric water content value in Equation 3.2 with the value estimated for the lower limit of readily available water, that is, the water held at a pressure potential of -1.0 bar (Table A2.3).

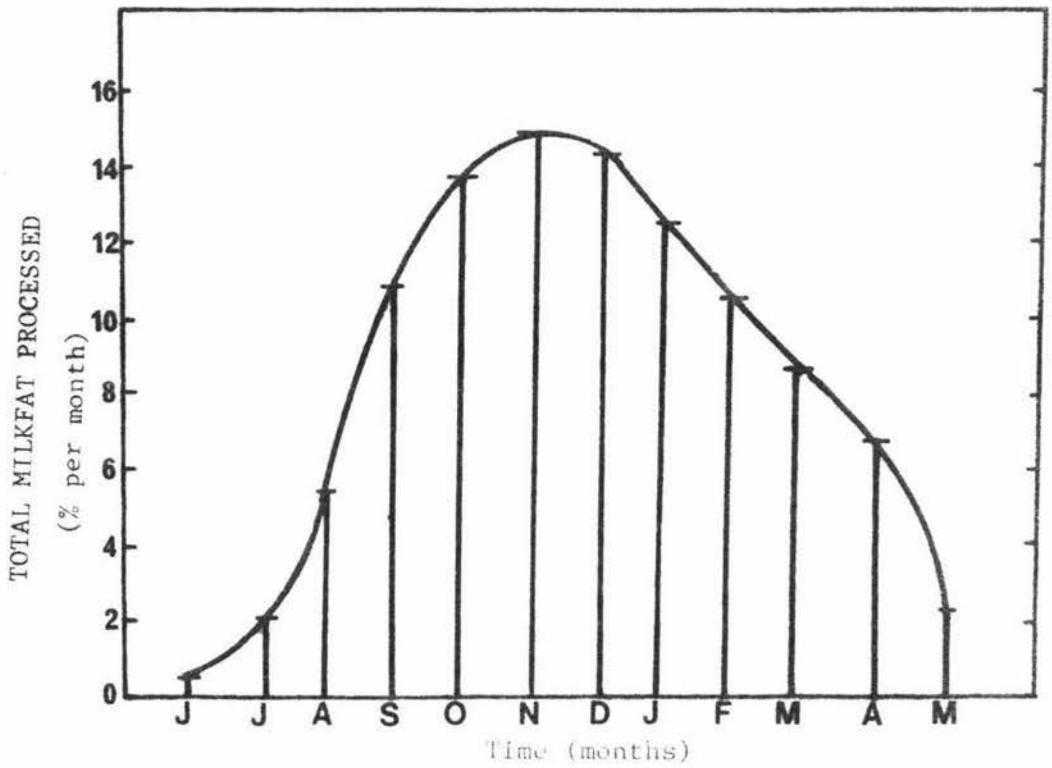

Fig. A2.1 Seasonal dairy production in New Zealand (averaged over the ten year period 1965-1975). After Boyer (1975).

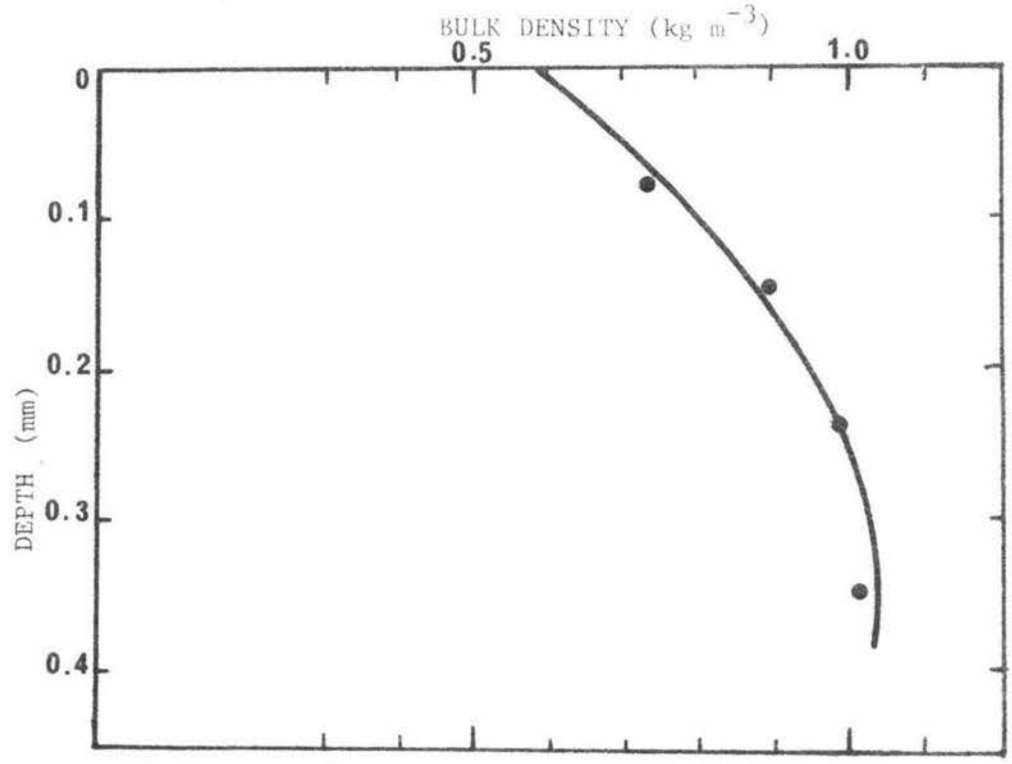

Fig. A2.2 Bulk density of the Te Rehunga disposal disposal site profile.

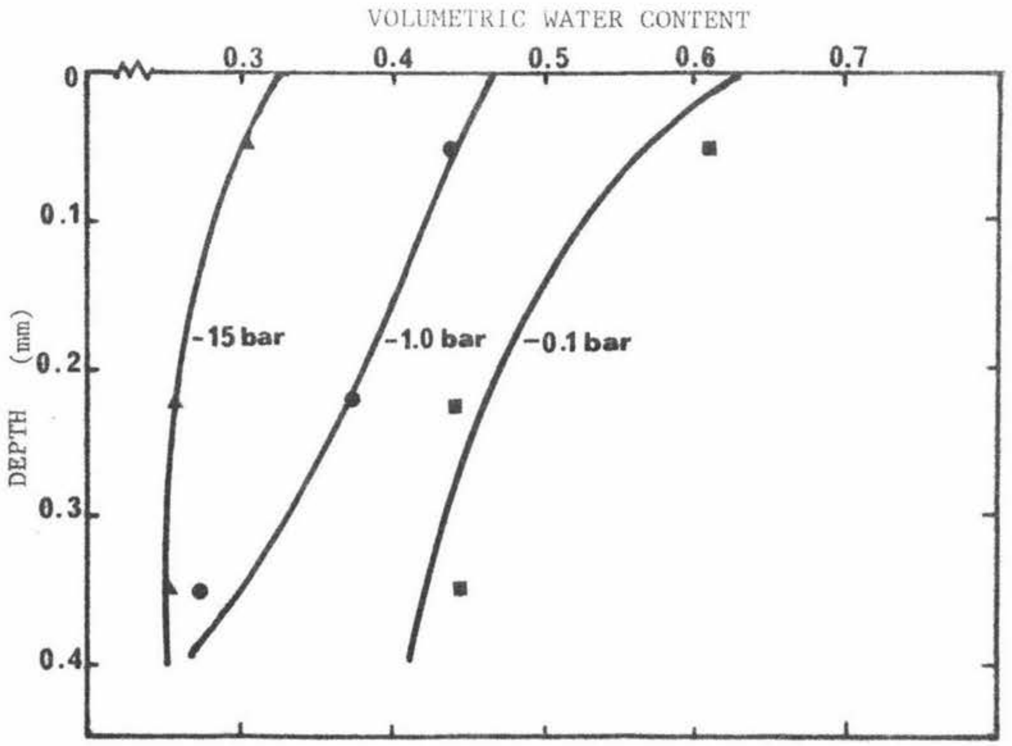

Fig. A2.3 Water retentivity characteristics of the Te Rehunga disposal profile.

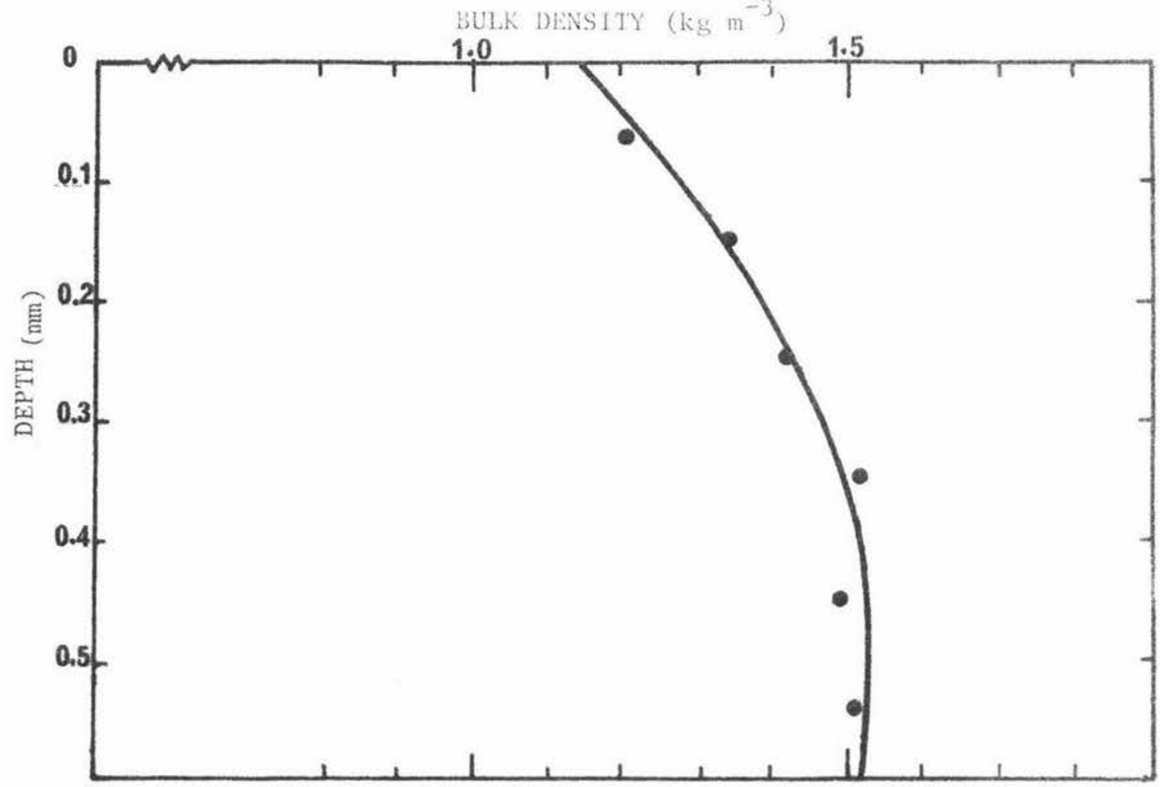

Fig. A2.4 Bulk density of the Longburn Disposal site profile.

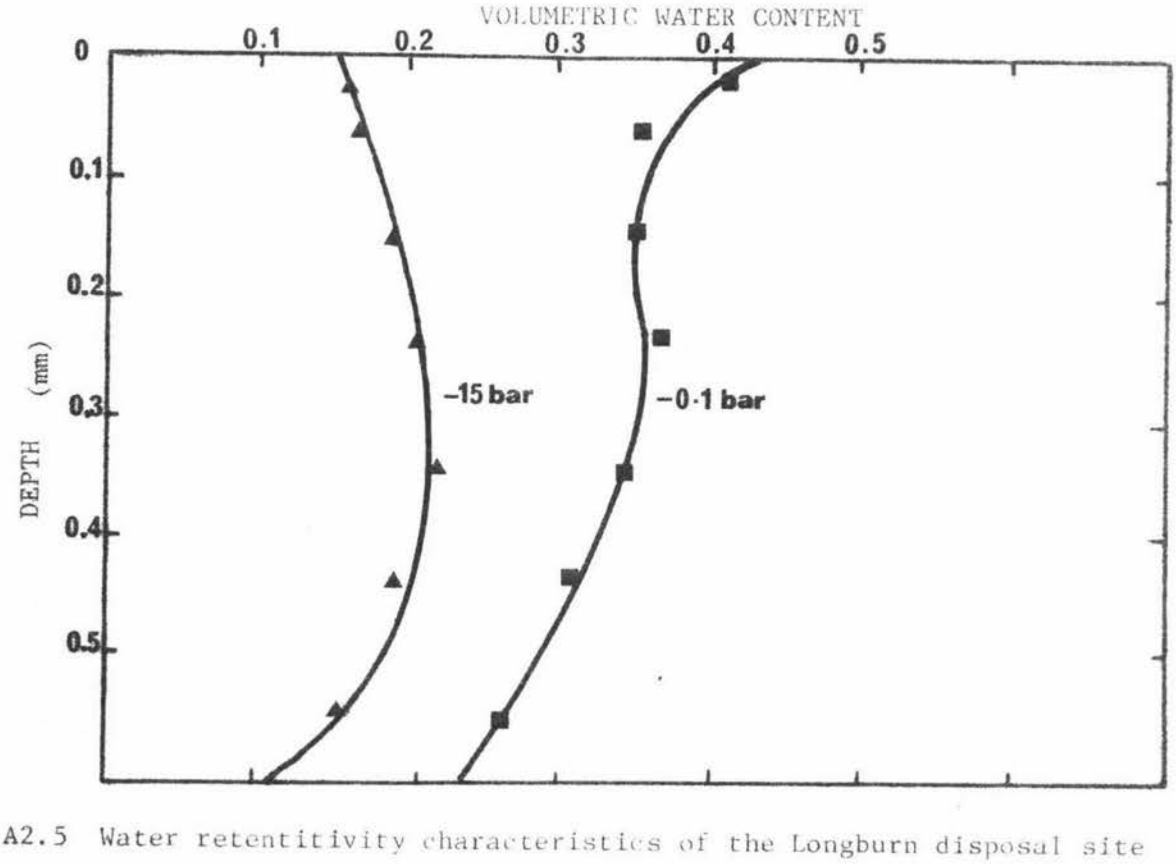

Fig. A2.5 Water retentitivity characteristics of the Longburn disposal site profile.

Table A2.4 An example of the working procedure involved in the construction of a daily water balance (from the Te Rehunga water balance)

Date		Evapotranspiration	Rainfall	Storage	Runoff	Deficit (mm)
1970 March	1	2.6	4.6	-64.1		
	2	2.6		-66.7		
	3	2.6		-69.3		
	4	2.6		-70		1.9
	5	2.6		-70		2.6
	6	2.6		-70		2.6
	7	2.6		-70		2.6
	8	2.6	4.3	-68.3		
	9	2.6		-70		0.9
	10	2.6		-70		2.6
	11	2.6		-70		2.6
	12	2.6		-70		2.6
	13	2.6	0.5	-70		2.1
	14	2.6	7.6	-65.0		
	15	2.6	0.5	-67.1		
	16	2.6	1.8	-68.9		
	17	2.6	17.5	-53.0		
	18	2.6	58.4	0	2.8	
	19	2.6	24.9	0	22.3	
	20	2.6	2.8	0	0.2	
	21	2.6	3.0	0	0.4	
	22	2.6		- 2.6		
	23	2.6		- 5.2		
	24	2.6		- 7.8		
	25	2.6	6.1	- 4.3		
	26	2.6	5.6	- 1.3		
	27	2.6	1.8	- 2.1		
	28	2.6		- 4.7		
	29	2.6		- 7.3		
	30	2.6		- 9.9		
	31	2.6	50.3	0	37.8	

Table A2.5 Maximum daily rainfall figures (in mm) for Te Rehunga (over the 10 year period 1967-1977) and for Palmerston North (over the 42 year period 1928-1970)

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
Te Rehunga	69	171	112	51	84	83	59	49	69	63	47	130	171
Palmerston North	82	92	85	69	74	49	59	51	46	37	47	63	92

APPENDIX III

DERIVATION OF THE RESPIROMETER EQUATION

The gas equation is

$$PV = nRT$$

where P is the pressure (Pa), V is the volume (m^3), n is the number of moles of gas, R is the gas constant (8.3J K^{-1} mol^{-1}) and T is the temperature (K). If

$$P_1V_1 = n_1RT$$

is used to describe the initial conditions at t1, and

$$P_2V_2 = n_2RT$$

represents the gas equation at time t2, then the number of moles of oxygen consumed over the period t2-t1 is given as

$$n_1-n_2 = (P_1V_1 - P_2V_2/RT).$$

The pressure change in the respirometer is given as

$$P_1-P_2 = \rho_w gh$$

where ρ_w is the density of the manometer fluid (kg m^{-3}), g is acceleration due to gravity (m sec^{-2}), and h is the change in height of the manometer fluid (m). The volume change in the respirometer is

$$V_1-V_2 = \pi r^2 h$$

where r is the radium of the manometer tubing (m). Hence the number of moles of oxygen consumed between t1 and t2 can then be determined as

$$n_1-n_2 = \left[P_1V_1 - (P_1-\rho_w gh)\ (V_1-\pi r^2 h)\right]\ /RT$$

APPENDIX IV

ADDITIONAL RESPIROMETRY STUDIES AND CARBOHYDRATE ANALYSIS DATA

A4.7 Respirometry Studies

Using the procedure described in Section 4.3.1, an experiment was designed to examine the effectiveness of sodium azide as a microbial inhibitor. Three treatments using whey effluent, effluent plus inhibitor and water were applied to surface layer cores from the control site at Te Rehunga. The treated cores were crumbled and sieved, then placed into the respirometer. As with all the respirometry experiments, a 45 minute equilibration period was allowed before oxygen consumption was recorded. Mean value changes for the two replicates used per treatment are shown in Fig. A4.1. Comparing the responses of the whey effluent and effluent plus inhibitor treatments it is seen that sodium azide did limit microbial activity, but that inhibition was not complete. A second treatment application, 24 hours after the initial application, again illustrated that sodium azide inhibition was incomplete.

A second respirometry experiment was carried out using soil samples from the depression areas at the Longburn disposal site (Section 7.2.2). Two treatments were employed in this experiment, one where the soil samples were placed in the respirometer vessel as large "undisturbed" aggregates, and the other where the soil was sieved prior to oxygen consumption measurement. Three replicates per treatment were used; the mean changes in oxygen consumption with time for replicates are shown in Fig. A4.2. The results obtained illustrate two major points. Firstly, a relatively high oxygen consumption is noted in both treatments, thus suggesting that a substantial amount of readily oxidisable carbohydrate material is stored in the depression area soils. Secondly, since the sieved soil exhibited the greater response, it would appear that the oxygen supply rate is the factor limiting soil microbial respiration in the depression area soils.

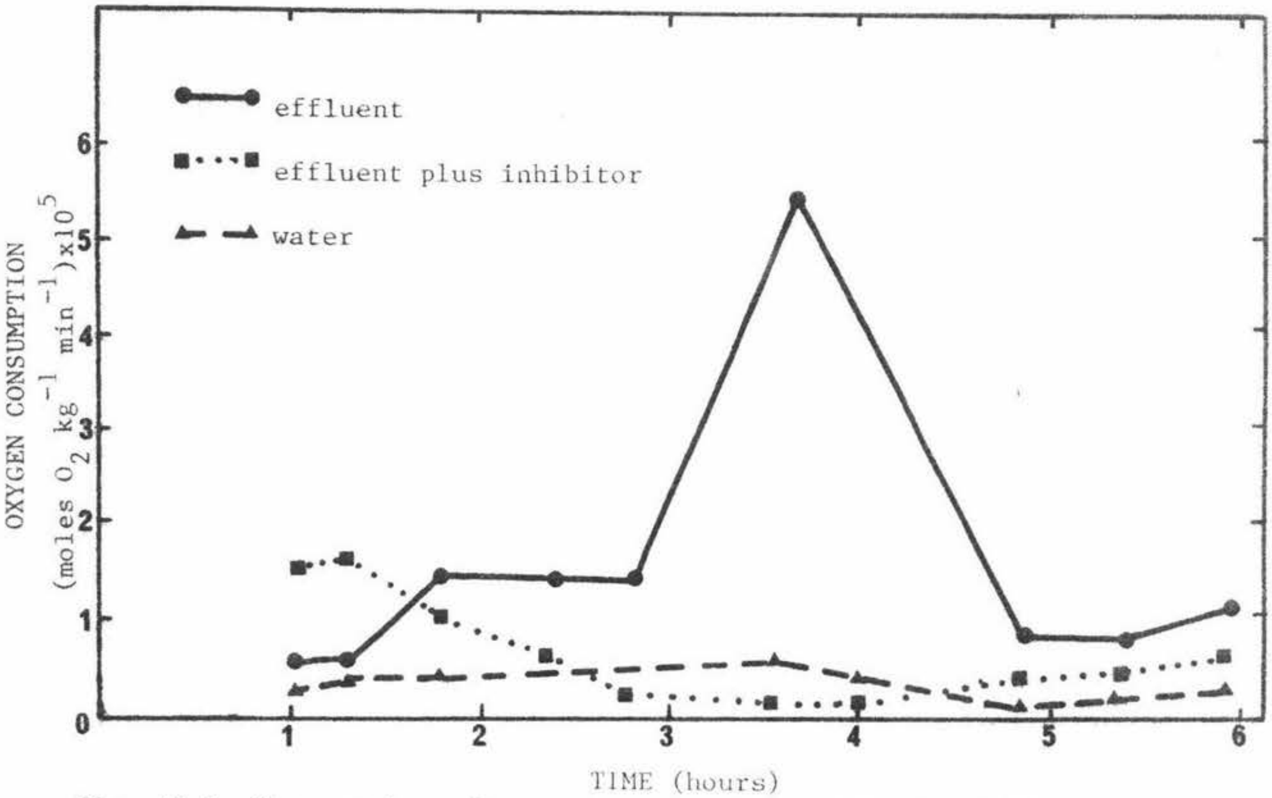

Fig. A4.1 Changes in soil oxygen consumption with time following the application of simulated effluent, effluent plus inhibitor and water treatments to soil samples from the Te Rehunga control site.

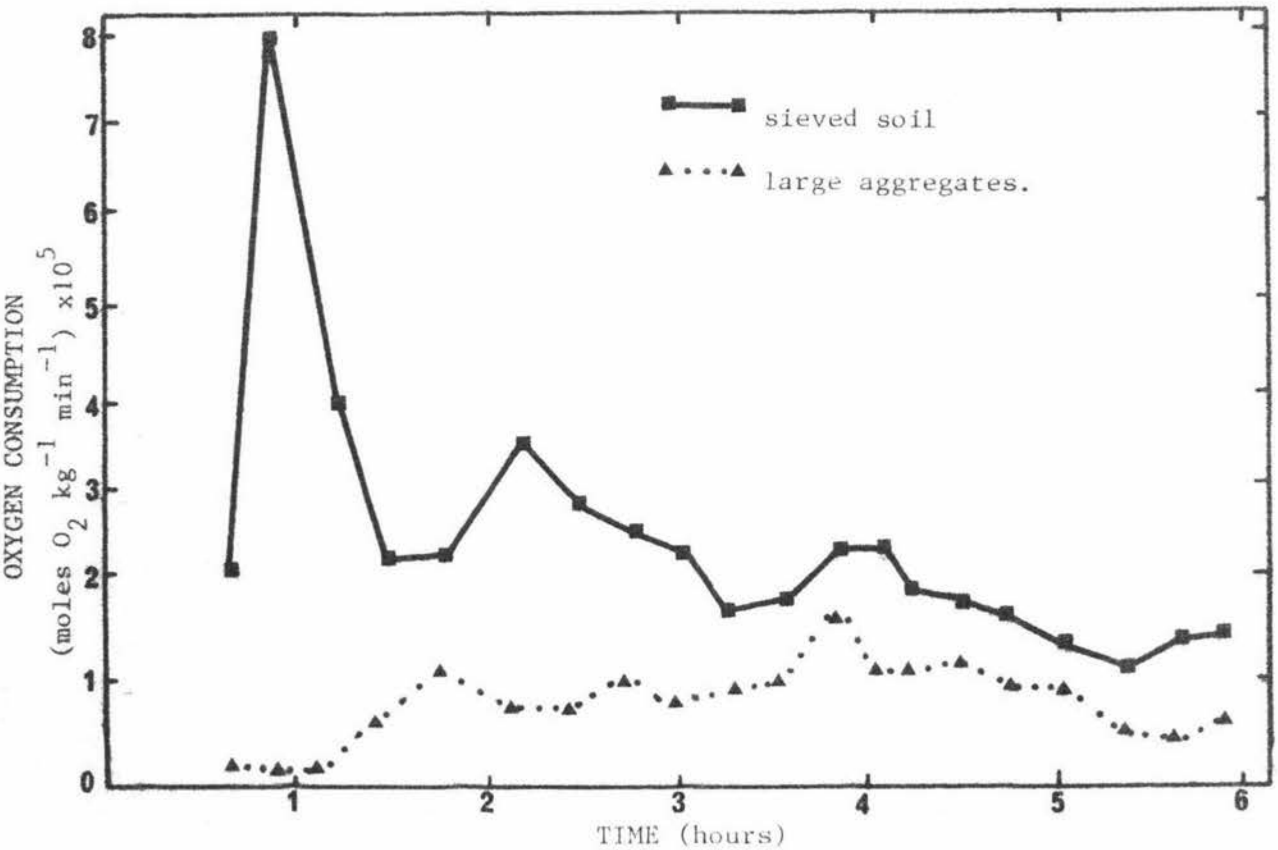

Fig. A4.2 Changes in soil oxygen consumption with time following either the sectioning or sieving of soil cores collected from the depression areas of the Longburn disposal site.

A further respirometry experiment was set up to examine soil microbial respiration changes following addition of whey effluent to surface layer soil samples from the Tokomaru and Longburn disposal sites. Mean value changes of the three replicates used per treatment are shown in Fig. A4.3 and A4.4. Results show that addition of whey effluent to both soils induced a marked and rapid increase in oxygen uptake, as was also evident for the Te Rehunga soil samples in Figs. 6.1 and 6.2.

A4.2 Carbohydrate Analysis

In conjunction with the laboratory studies of hydraulic conductivity (Section 5.4), analyses were made to determine both the amount of total carbohydrate material applied to the core in the whey effluent, and the amount lost from the core through leaching; the method used is described in Section 3.4.3. Once the carbohydrate inputs and losses for each core were found it was possible to calculate the proportion of any single dose of whey effluent carbohydrate which is either retained by, or leached through, the soil core. A summary of results for the different soils used is presented in Table A4.1. Since the proportion of carbohydrate retained or leached appears to be largely determined by the K value, individual core values for this parameter are also presented.

Carbohydrate analysis of the leachate from the effluent plus inhibitor treatment cores used in Section 5.4.3 showed that the inhibitor employed (sodium azide at $100 \mu g\ g^{-1}$) was not totally effective as a microbial inhibitor. This analysis showed that the effluent plus inhibitor treatment cores consistently removed a high proportion of the carbohydrate applied. With one soil group (MSL0-100mm) the effluent plus inhibitor treatment cores were found to remove as much of the applied carbohydrate as did cores not receiving the inhibitor.

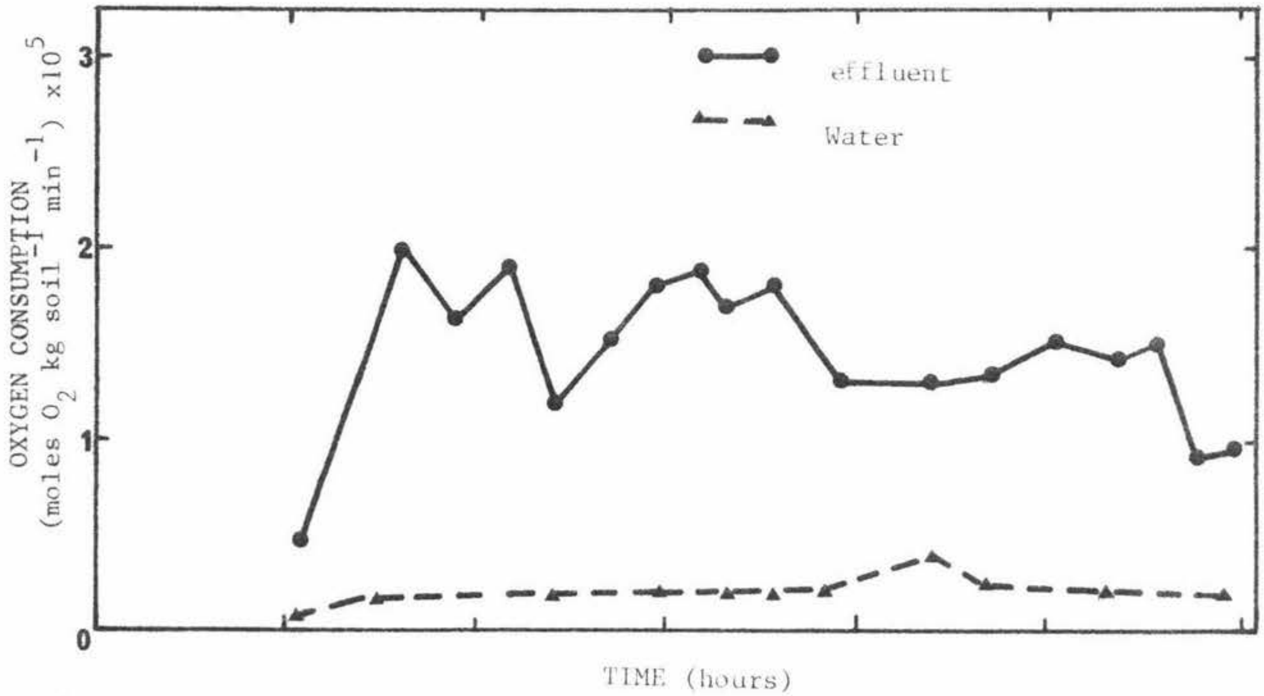

Fig. A4.3 Changes in soil oxygen consumption with time following application of simulated effluent or water to surface layer soil from the Tokomaru disposal site.

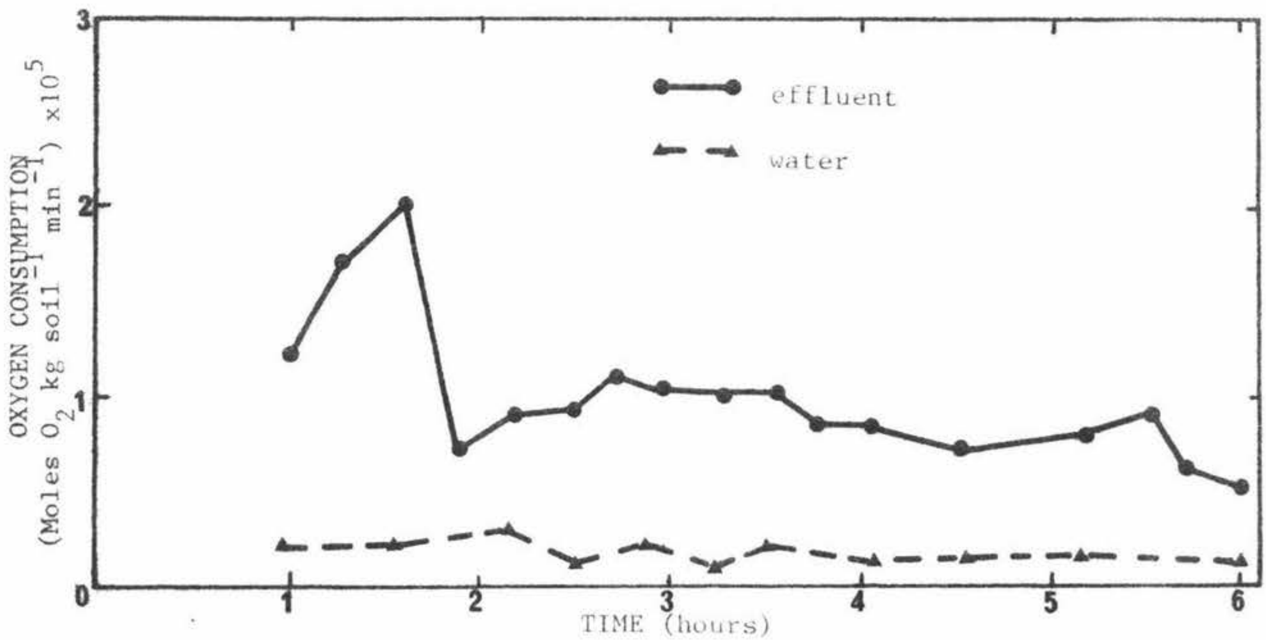

Fig. A4.4 Changes in soil oxygen consumption with time following an application of simulated effluent or water to surface layer soil from the Longburn disposal site.

Table A4.1 The percentage of applied carbohydrate collected in the leachate following addition of a 35mm dose of simulated effluent to "undisturbed" soil cores

Soil	Replicates							
	1	2	3	4	5	6	7	8
TSL (0-100mm)								
K value (mm day^{-1})	1890	1660	1475	1150	395	344	230	177
% leached	53	43	40	28	31	9	20	3
TSL (100-200mm)								
K value (mm day^{-1})	681	563	180	177	160	148		
% leached	30	53	43	18	37	52		
KSL (0-100mm)								
K value (mm day^{-1})	2000	1920	1040	840	274	210	50	31
% leached	41	55	41	38	28	37	13	11
KSL (100-200mm)								
K value (mm day^{-1})	3510	1900	1150	1070	813	640		
% leached	67	62	67	26	38	32		
MSL (0-100mm)								
K value (mm day^{-1})	1920	890	560	285	217	180	175	
% leached	56	46	38	54	33	22	52	

APPENDIX V

ANALYSIS OF EFFLUENT SAMPLES

Effluent samples were collected from the sprinkler line at the Longburn dairy factory. Samples were analysed in the laboratory for total and suspended solids, electrical conductivity and pH, using the procedures described in Section 3.4. A summary of the results is presented in Table A5.1.

Table A5.1 Composition analysis of effluent samples from the Longburn dairy factory

Date Collected	Total Solids (%)	Suspended Solids (%)	Electrical Conductivity ($S\ m^{-1}$)	pH
15/5/78	0.66	0.10	0.125	4.65
17/5/78	0.83	0.35	0.115	5.3
19/5/78	0.50	0.33	-	-
23/5/78	1.30	0.17	0.027	6.3
2/6/78	0.64	0.25	0.128	5.4
8/6/78	0.76	0.10	0.160	5.0
10/6/78	0.80	0.15	0.170	5.15
Simulated effluent	0.58	0.09	0.064	-

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