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SUBJECT **Lichfield site irrigation - Assessment of Environmental effects**

CONTENTS

This report has been prepared from the information obtained from the following sources:

1. *Lichfield Dairy Processing Plant - Assessment of environmental effects, prepared by NZ Dairy Group of Companies, June 1994.*
2. *Soil Data - prepared by M McLeod, Landcare Research, 13 July 1994.*
3. *Anchor Products Ltd Lichfield site, Management Plan for the Spray Irrigation of Wastewater, prepared by JW Barnett and JM Russel, July 1994.*
4. *Lichfield Dairy Processing Plant - Assessment of environmental effects (Supplementary Information), prepared by NZ Dairy Group of Companies, July 1994.*
5. *Buxton Farm Irrigation - Assessment of environmental effects, prepared by N Selvarajah, May 1994.*
6. *Buxton Farm Wastewater Irrigation Project - Evidence for Joint Hearing, prepared by Anchor Products, May 1994.*
7. *MIRINZ report (by JM Russel and SB Lindsey) for DRI on denitrification rates from dairy factory waste applied onto pasture, February 1990.*
8. *Environment Waikato databases.*
9. *Various information received through fax from Kingett Mitchell & Associates Ltd.*
10. *Other references (see reference section).*

1. INTRODUCTION

This report assesses the reliability of the prediction of the effects on soil and ground water due to the proposed land disposal of waste water produced from the proposed Anchor Products dairy factory at Lichfield. The report also develops environmental indicators and monitoring programmes for the key waste water characteristics and soil and ground water quality.

The keys issues are identified as (a) hydraulic loading (b) nitrogen loading and (c) sodium loading.

2. HYDRAULIC LOADING

Objective: *Maintaining a hydraulic loading which does not enhance nutrient leaching, surface ponding and surface runoff.*

2.1 Average daily volume of effluent

The volume of waste water applied onto land can vary greatly and hence use of an average annual or daily value can often be misleading. The 20 day rolling average volume of waste water that is anticipated is 4100 m^3/day for a 200 day peak production period. Considering the proposed annual N loading rate of 400 kg N/ha, the approximate average daily flow is estimated as 3050 m^3/day for the milking season (270 days). Generally maximum volumes of waste water are applied onto land during spring and summer. Waste water will be sprayed for a maximum of 8 hours over a 48 hour period hence approximately 50 mm is applied per application.

Assumption:	
Number of days irrigated	= 270 days
Data on hydraulic loading:	
Maximum application rate	= 6 mm/hour
Area available	= 164 ha
Proposed volume of irrigation	= 3050 m^3/day
Annual loading	= 823500 m^3
Volume sprayed per area	= 5021 m^3/ha/year
Depth of application	= 502 mm/year
Application per dose	= 50 mm

2.2 Suitability of the site for irrigation

The irrigation site comprises a rolling landscape with varying soil depth (from <500 mm to >2000 mm). Soil depth is influenced by underlying welded or unwelded ignimbrite rock material. The soil type is Taupo silt loam which typically has low bulk density and high porosity. The soil also has varying depth of A horizon due mainly to apparent land disturbance. Due to the rolling landscape, high soil variation and adverse climatic conditions, the Lichfield site is markedly different to the Hautapu irrigation site and hence Lichfield is a difficult site for waste water irrigation management and planning. Greater frost incidence, high rainfall (1500 mm) and slope can make the irrigation more difficult where irrigation has to be performed daily regardless of weather or soil conditions. This is because the cheese factory waste water has poor storage quality due to rapid fermentation, which makes the waste water extremely acidic and hence unsuitable for irrigating onto pasture. The combined effect of net rainfall and an additional 502 mm hydraulic loading due to irrigation with high sodium loading, is likely to cause some drainage problems at the proposed disposal site. Moreover, it is not known whether the percolated water is likely to perch or how rapidly the percolated waste water can move through the underlying ignimbrite rock. Excess build-up of percolated waste water can result either in poor soil conditions for pasture growth or subsurface accumulation and surface runoff. With the existing information, however, it is difficult to predict such effects.

Similarly, it is felt that the amount and the quality of information available regarding the receiving environment for determining a suitable nutrient loading rate is very poor. Frequent reference has been made regarding a shallow aquifer at 2-3 m depth, although the characteristics of this aquifer (e.g. thickness, throughflow and areal extent) are not known. What is the interaction of this aquifer with other underlying aquifers and the Pokaiwhenua stream? What is the ground flow direction? These questions and many more have made the environmental impact assessment extremely difficult. It is emphasised that under such circumstances the only safe approach is to adopt conservative (cautious) estimates to determine waste water application rates.

2.3 Application rate

Although the proposed application rate (6 mm/hour) is considered as suitable for most soil types under flat conditions, surface runoff of the irrigated waste water can occur for undulating to gently rolling land. Runoff problems can be more severe during heavy rainfall or when soils are hydrophobic due to dry soil surface conditions. It is thus recommended that the irrigation system installed at the Lichfield site should be able to be operated at a range of application rates such as 2-6 mm/hour to suit the soil and weather conditions encountered at the time of application.

The Management Plan (Anchor Products) acknowledges that there will be ponding and subsequent ripping of soil and/or undersowing of grass. Proper land disposal systems should not allow any surface ponding. Frequent surface ponding from irrigated waste water enriched with high carbonaceous material can cause anoxic conditions in soil, which is detrimental for soil and pasture. Surface ponding is unlikely on hill soils due to the slope (surface ponding on hill soils will result in overland runoff of effluent) whilst valley soils are more susceptible to ponding.

2.4 Net hydraulic loading

It appears that there will be little or no irrigation during May, June and July which coincides with the typical regional non-milking season. From long-term rainfall records (1951 to 1980) available at Lichfield (NZMS, 1984), average rainfall during the balance of the year is estimated as 1100 mm, which is 73% of the average annual rainfall (1500 mm). According to this estimation, the daily rainfall equates to 4 mm/day. The average evapotranspiration rate (ET) for the irrigation period is approximately 2.6 mm/day (700 mm for the irrigation period when annual evapotranspiration is 750 mm). These data indicate that there is a rainwater surplus of 400 mm/year or 1.5 mm/day during the irrigation period which in turn demonstrates that any additional hydraulic loading through land application of effluent will contribute to either surface runoff or ponding, or recharge of the shallow aquifer. This additional hydraulic loading in the spray area is 502 mm/year. Overall, the site will experience a net annual hydraulic loading of 1252 mm which is 25% greater than that of the Hautapu (Bardowie) site.

Table 1. Estimation of net hydraulic loading for the irrigation period (270 days)

	Daily	Annual
Av. rainfall	4.0 mm	1100 mm
Av. ET	2.6 mm	700 mm
Application rate	1.9 mm	502 mm

Net hydraulic loading	3.3 mm	902 mm

Use of precipitation index (PI)
A precipitation index has often been used to minimise or avoid any potential surface runoff or ponding of effluent applied onto land:

$$PI = (0.2 \times R_4) + (0.5 \times R_3) + (1.0 \times R_2) + (1.5 \times R_1)$$

where R_1, R_2, R_3 and R_4 is the rainfall (mm) on the previous 4 days respectively.

During wet periods PI can be used effectively to manage net hydraulic loading. The conventional engineering approach to waste water irrigation is to use land as a `filtering system' hence it encourages greater hydraulic loading on well drained soils. Thus the PI for well drained soil is taken to be several folds higher than that for poorly drained soils. Such an approach accounts for surface runoff, but ignores nutrient leaching from top soil.

2.5 Effect of hydraulic loading on nutrient leaching

Nutrient leaching is a dynamic and cumulative process which increases with hydraulic loading. For a given hydraulic load the extent of leaching varies according to the soil type, soil moisture conditions and the nutrient content of irrigation water (Scotter, 1993). Irrigating waste waters have more deleterious effects on the receiving environment than irrigating pure water because dissolved nutrients can move with waste water in the soil profile.

When waste water containing nutrients is irrigated onto dry soil the nutrients are absorbed into small soil pores and the potential nutrient leaching is reduced even if there is a high rainfall following irrigation. However, when waste water is applied onto wet soil, the nutrients are absorbed mainly in large pores resulting in a high leaching potential (Tillman *et al.*, 1991). Thus a high rainfall event following irrigation can leach most of the nutrients that are located in the macropores. It is emphasised here that proper use of PI will only prevent surface runoff, but it does not guarantee minimal leaching loss. This is because PI considers only the preceding rainfall for 4 days and it does not account for post irrigation rainfall. Consequently, any good waste water irrigation practice should adopt conservative hydraulic loading rates. Irrigation should be planned according to the season and the ability for soils to assimilate the applied nutrients.

The proposed irrigation rate is 50 mm over a two day period. Assuming a maximum soil pore volume of 70%, the leaching depth will be 71 mm (i.e. about 70 mm depth is available in every 100 mm soil column for the water to occupy). If the top 100 mm soil is fully saturated then the leaching depth will be at least 171 mm (i.e. assuming little or no irrigated water is held in the 100 mm depth saturated soil column and the balance of the 100 mm `dry' soil column has 70% pore volume). If the top 200 mm soil is saturated, the leaching depth will be at least 271 mm.

Any waste water irrigation applied onto grazed pasture should attempt to confine the applied waste water nutrients and the existing soil nutrients within the top 200 mm soil where most pasture roots are located. Plant uptake of nutrients is used as the major nutrient removal mechanism and hence nutrient leaching below the root zone has a greater potential to contaminate ground water. Thus the proposed hydraulic loading is acceptable when little or no rainfall is received prior to irrigation. Under wet conditions, however, such a rate of application can cause nitrate leaching. It is recommended that either the rate of application should be halved (i.e. 25 mm per dose) or a PI of 20 should be used during August, September and October.

Nitrate has a greater potential to be reduced to gaseous forms in the top soil due to the presence of high organic carbon and bacteria. High hydraulic loadings can enhance anoxic conditions due to saturation, and increase the potential for denitrification of nitrate into gaseous forms. This process requires greater residence time for nitrate in the top soil. However, high hydraulic loadings can also flush down a high proportion of nitrate present in top soil and thus reduce the nitrate available for denitrification. Organic carbon in soil (even in dissolved forms) is not as mobile as nitrate and hence nitrate escapes from the zone of denitrification and becomes available for ground water contamination. Moreover, considering the irrigated area will be used for grazing, further nitrate (and urea-N) leaching will occur from urine voided by dairy cows.

According to the hydraulic loading of 50 mm per application, the amount of N loading per application will be 40 kg/ha. This estimate has been made from the average effluent total N content of 80 g/m^3. Considering the instantaneous N loading, this loading rate is considered to be agronomically acceptable. At this rate of application each paddock will be irrigated at least 7 times during the season (@ 300 kg N/ha/year). As suggested earlier, at 25 mm/dose irrigation can be performed 15 times a year per paddock. Alternatively, using PI, a higher PI can be used with a lower application rate as long as the net hydraulic loading remains constant (i.e. 50 mm) (e.g. Table 3). If irrigation has to be performed when PI > 20 mm, hydraulic loading can be estimated by subtracting PI from net hydraulic loading.

Table 2. Different combination of net hydraulic loading parameters

PI	Hydraulic loading	Net Hydraulic loading
25 mm	25 mm/application	50 mm/application
30 mm	20 mm/application	50 mm/application
40 mm	10 mm/application	50 mm/application

Combining a higher PI with lower irrigation application rate will help to reduce the loading on the disposal area during wet weather conditions.

It becomes clear that hydraulic loading is an integral part of effective land treatment systems. Thus every effort should be made to minimise the volume of waste water produced in the factory.

3. NITROGEN LOADING

Objective: *Avoid or minimise soil and water contamination whilst maintaining or enhancing soil and pasture quality*.

The waste water generated at the proposed cheese factory is a good source of essential plant nutrients. It contains a considerable amount of macronutrients such as N, P, K, Ca, Mg, and S. Thus land disposal of this waste water is considered very useful for pasture production. The waste water also contains a substantial amount of Na and the land disposal of this element is addressed in the latter part of the discussion.

3.1 Nitrate contamination

The presence of nitrate in ground water provides an indication of ground water contamination. According to the New Zealand drinking water standards the maximum acceptable level for nitrate-N is 10 g/m^3 (Board of Health, 1989). Bottle fed infants less than 6 months old consuming water containing nitrate are reported to have developed a disease called methaemoglobinaemia (`blue baby' syndrome). Overseas studies report many such cases with several cases resulting in death (Winton et al., 1974). To date, no cases have been reported in New Zealand, although methaemoglobinaemia is not classified as a notifiable disease by the New Zealand Health Department. However, the symptoms for the Sudden Infant Death Syndrome (`cot death') are similar to that of `blue baby' syndrome, implying that there may have been methaemoglobinaemia cases in New Zealand which have never been noticed. In adults consumption of drinking water with high nitrate levels have been linked to gastric cancer and hypertension cases (quoted by Burden, 1982).

Apart from being a potential health hazard, due to subsurface flow of ground water into streams or rivers, nitrate in ground water can pollute waterways causing algal blooms and may subsequently affect aquatic life such as fish. Many waterways in the region are used for recreation and unwanted algal growths can affect the revenue gained by tourism. High nitrate flow into the sea combined with phosphorus availability has also been considered as one of the main factors for toxic algal blooms reported frequently around the globe and recently in New Zealand. Moreover, if the environment is degraded, unnecessary trade barriers can be imposed by overseas trading partners on the export of food products from New Zealand. Currently, many European trading partners are spending billions of dollars in the management of their environment, hence the high cost of production. In New Zealand the cost of production is relatively low, principally because there is an `environmental subsidy' attached to the total cost of production (i.e. the cost of environmental degradation is not considered in the production of export food products). Many European trading partners believe that countries which trade with the EC should maintain similar environmental standards, hence a `realistic' cost of production. It should also be emphasised that New Zealand has a good marketing potential because of its clean and green image overseas. Every effort should be made to maintain this image, because once such an image is lost it is difficult to regain. One of the major problems with nitrate is that contaminated ground water is difficult to clean up.

3.2 Nitrogen in a grazed pasture system

Efficient N management is very important in soils receiving effluent with a high N content. Poor N management will lead to substantial ground water pollution. When the input exceeds the output of N, a surplus of N in soils occurs and hence potential for leaching loss of applied N. It is essential to use N transformation models to predict the fate of applied N. A proper model will consider the important N transformation processes in the system. Once the model is derived a loading rate can be determined. A pasture system with zero grazing which is frequently referred to as `cut and carry' is easy to manage and requires only a simple N model since crop uptake is the major N removal mechanism. For example, in order to produce 16000 kg dry matter per ha with 4% N will require 640 kg/ha/year. Assuming that the soil has very low levels of organic-N, for a pure ryegrass pasture system approximately 650 kg N/ha/year can be applied in several split doses throughout the growing season, causing little or no ground water pollution. However, under a grazed pasture system N ingested by animals is recycled through animal excreta without being transferred offsite as in a `cut and carry' system. Consequently, the conventional N loading rate determination based mainly on the amount of crop uptake and N removal in animal products is considered to be inappropriate for a grazed system. Moreover, under a grazed pasture system, whilst high soil N build-up leads to high nitrate leaching losses, high N content in pasture leads to greater N ingestion by grazing animals and consequently higher excreta-N loss through urine (Jarvis et al., 1989). It has been well documented that the presence of grazing animals is the driving force for nitrate leaching in soil (Selvarajah, 1994). This is because cow urine spots contain up to 1000 kg N/ha. Although it is well known that increasing N application results in increased dry matter production, the resultant increased stocking rate or high plant N content can lead to greater N loss through leaching from the system.

3.3 Assessment of proposed nitrogen balance

3.3.1 Products removal of nitrogen

According to the information provided by Anchor Products, milk fat production in the Tokoroa area with a drymatter production of 14000 kg/ha/year is greater (i.e. 650 kg milk fat/ha/year) than other areas in the Waikato region. Assuming a milk protein production of 500 kg/ha/year for the corresponding milk fat production, the N removed through milk production should be 80 kg/ha/year. Further calculations indicate that the above drymatter production is not sufficient to produce such a high milk yield. High milk yields can be obtained through a increased supplementary diet or increased fertiliser application rates. Most farms in the Tokoroa area adopt both these practices. Although the dairy factory waste water contains essential nutrients in abundance, the pasture production or milk production may not be increased substantially. High loading of this waste water can reduce milk production due to poor pasture performance. Such a reduction has been observed at the Hautapu irrigation site (Bardowie farm). Despite a high N loading rate (1100 kg/ha/year), the maximum milk fat yield ever obtained at this site was only 619 kg/ha/year (minimum being 504 kg/ha/year). Various factors could have attributed to such a low yield, among which the major factor is the poor pasture performance due to water logged conditions which was caused by the high hydraulic loading combined with heavy sodium loading at this site.

In short, real benefit for pasture and soil from the use of dairy factory waste water can only be obtained when waste water is used in moderate amounts. However, it can be argued that since the primary objective of the project is waste treatment rather than farm production, the reduced yield due to waste water application should not be of concern, considering there is little or no environmental impact. Without knowing the past average production capacity of the three dairy farms at Lichfield, it is difficult to comment on the potential reduction or increase in milk yield due to waste water application. However, considering the high net hydraulic and sodium loadings, an average milk fat production of 500 kg/ha/year is anticipated, which translates to 385 kg protein/ha/year (62 kg N/ha/year). If loss of N due to replacement stock is approximately 10 kg N/ha/year, the total amount of N removed through animal production will be 72 kg/ha/year. The estimate provided by Anchor Products was 65-75 kg N/ha/year, and therefore is considered to be an appropriate estimate for N removed through production for the Lichfield site receiving cheese factory waste water at a N loading rate of 400 kg N/ha/year. However, at a lower loading rate such as 300 kg N/ha/year, there will be associated low hydraulic and sodium loadings (a reduction of 25%) and hence greater pasture performance. It is anticipated that the average milk fat production at 300 kg N/ha/year loading rate will be 600 kg milk fat/ha/year which will remove 75 kg N/ha/year (462 kg protein/ha/year). Thus the total N removed through animal products will be 85 kg N/ha/year.

3.3.2 Ammonia volatilisation

Ammonia volatilisation occurs when ammonical-N is present in soil at excessive levels. Losses sustained can vary according to the environmental conditions (rainfall, temperature, and soil moisture), agronomic factors (fertiliser-N application, liming and presence of grazing animals), soil types and presence of plant cover. Assuming that protein is the only form of N present in the waste water, the potential for instantaneous ammonia loss will be zero. The extreme pH values provided for the waste water suggest that at times pH levels reach up to 11. Under such conditions some protein-N can be hydrolysed into ammoniacal-N form. However, considering the carbonaceous nature of the waste water the potential for ammonia loss can be suppressed through rapid immobilisation, whereby ammoniacal-N is used for biosynthesis by heterotrophic bacteria. Such a process requires only the ammoniacal-N form, and nitrate-N is not consumed by these bacteria for the immobilisation process (Wickramasinghe *et al*., 1985).

In the long-term, however, ammoniacal-N can be released from mineralisation of applied protein and the urine voided by cows in which only urine patches have greater potential for ammonia loss. A recent study on ammonia loss from urea-N applied to a wide range of soil types in New Zealand demonstrated that volcanic ashes have very low potential for ammonia loss (Selvarajah *et al*., 1993). Moreover, volatilisation can be completely suppressed during an irrigation or rainfall event due to urea-N and ammonical-N leaching (Selvarajah, 1991). The presence of pasture cover can absorb a substantial amount of ammonia volatilised (Lockyer and Whitehead, 1987). Under the site field conditions and the proposed irrigation conditions all the above factors favour very low ammonia volatilisation. Extrapolating the field ammonia loss from urea broadcast to pasture under Canterbury spring conditions using micrometerological methods, my estimation of ammonia loss for the proposed pastoral system will be up to 5% of the urine-N deposited. If 720 kg N/ha/year is assumed as herbage-N intake by cows, after the removal of products the remaining N will be animal excreta (i.e. 648 kg N/ha/year). Assuming 20 kg/ha/year of this excreta is deposited in the milk shed (when the total grazed area is 270 ha and 20 g N/cow/day is deposited from 915 cows for 270 days) the remaining 628 kg N/ha/year will contain 440 kg N/ha/year as urea-N (approximately 70% of the excreta is urea-N). An ammonia loss of 5% of 440 kg will be 22 kg N/ha/year. The ammonia loss estimated by Anchor Products is 25 kg N/ha/year and this estimate is considered to be appropriate under the given conditions.

3.3.3 Denitrification

(i) Denitrification in soil

Denitrification can occur through chemical and biological processes during which nitrite-N or nitrate-N is reduced to gaseous forms of N (nitrous oxide or dinitrogen). Unlike biodenitrification, the chemodenitrification process does not depend directly on microorganisms and occurs only when *nitrite-N* accumulates in high amounts. Many papers indicate that nitrite accumulation is rarely noticed in most soils and hence chemodenitrification is considered to be minimal under most conditions (Chalk and Smith, 1983). Since biodenitrification is the major pathway for nitrous oxide or nitrogen loss from the system, in this memo biodenitrification is referred to as denitrification.

The denitrification process requires four essential factors, meaning that in the absence of one of these factors there will be no denitrification loss. These factors are (a) availability of nitrate (b) absence of oxygen i.e. anaerobic conditions (c) an electron donor (e.g. available organic carbon) and (d) denitrifying bacteria. Other factors that can influence the extent of the denitrification process are moisture and temperature. Apart from moisture, temperature is one of the key environmental factors regulating the extent of denitrification losses. Consequently, denitrification losses are low at night and during cold seasons. Thus caution must be taken in extrapolating day time denitrification losses for the entire day and in extrapolating summer/spring measurements for autumn/winter conditions.

The sources of nitrate under the proposed irrigation practice will be nitrate released from protein applied through waste water irrigation, plant decay, and urine and dung deposited by animals. Unlike the Hautapu site, waste water will not contain any nitrate due to the combined use of phosphoric and sulphuric acid as a substitute for nitric acid.

(ii) Soil denitrification rates estimates

Very little information is available on the extent of denitrification losses from dairy factory waste water applied onto grazed pasture. The only field trial performed at the Bardowie irrigation site indicated that denitrification loss was not as much as anticipated from a land treatment system receiving high dissolved carbon (as lactose) and nitrate (as nitric acid) in the waste water (Russel and Lindsey, 1990). According to the estimate the loss was 110 kg N/ha/year of the 1490 kg N/ha/year applied which was only 8% of the applied-N. The nitrate sources for denitrification are (a) nitrate in the waste water and (b) nitrate present in soil. Unfortunately, the workers did not characterise the waste water for its nitrate levels and hence it is difficult to ascertain the amount of waste water nitrate that was responsible for denitrification loss measured in the field.

The study was conducted during the day time and gaseous samples were collected soon after the irrigation. Thus even a loss of 8% of applied-N may be an overestimate. The authors also observed that the gaseous losses dropped to background levels within 24 hours and that the losses recorded during a cold season were much lower. Russel and Lindsey (1990) also noted the relatively higher denitrification losses were sustained during dairy factory waste water irrigation onto an alkaline soil (pH 7.0) at Bardowie farm compared with meat processing plant effluent irrigation onto an acid soil (pH 4.9). This difference is due to the greater soil pH, a longer irrigation event, a greater C:N ratio and the presence of dairy cows at the Bardowie site. The average top soil pH for the Lichfield farm site is 5.6 and the denitrification potential could be much lower than at the Bardowie site.

Despite limited information available on denitrification losses from dairy factory waste water applied onto grazed pasture, the losses predicted by Anchor Products are 150-220 kg N/ha/year for the Lichfield site. This amounts to 38-55% of the applied-N, which is considered to be erroneously high for the system proposed. Bearing in mind the lack of information available at the Lichfield site on the receiving environment and the potential for denitrification losses it is extremely risky to rely on such estimates which contribute substantially to the N budget for the system; high denitrification estimates will lead to low nitrate leaching estimates and hence underestimate risk for ground water pollution.

The supplementary report provided by DRI argues that a substantial amount of denitrification losses can occur when such process is enhanced under favourable conditions. I agree with this statement fully because denitrification losses can be enhanced under anoxic conditions where there is sufficient supply of nitrate-N and available-C. The question is whether such conditions exist in a grazed system where high volumes of waste water are applied. This is because grazed systems require a continuous supply of grass, which will be affected heavily by anoxic conditions. The following points are worthwhile noting:

(a) As indicated in the report (refer Knowles, 1982), a substantial amount of gaseous N loss can be sustained at feedlots. Most of these losses occur through NH_3 volatilisation. This is obvious since urea-N in urine voided (which has >75% of the N excreted) will be rapidly hydrolysed to NH_3, which is susceptible for loss due to the alkaline soil conditions (often soil pH reaches up to 9.0). However, the extent of denitrification loss from feedlots was not clear from this reference.

(b) High denitrification losses can occur in soils irrigated with meat processing waste water. However, this can vary substantially from 2% to 30% of the applied-N (quoted by Russel and Cooper, 1992).

(c) As quoted (refer Barkle *et al*., 1993) high amounts of denitrification can occur in poorly drained soils with a controlled perched water table. The soil used in this study is gleyed (Te Kowhai silt loam) and has high denitrification potential due to water saturation and presence of reduced iron. However, this situation does not occur at the Lichfield site, because the Taupo silt loam is relatively well drained and aerobic.

(d) As quoted in the DRI report, Selvarajah *et al*. (1994) did indicate that a considerable amount of N can be lost through denitrification from soils irrigated with dairy factory waste water due to high C and hydraulic loadings. These workers, however, emphasised that even under such conditions maximum denitrification loss is *unlikely* to exceed 50% of the applied nitrogen. The DRI report misquoted that "...."a conservative loss up to 50% of the applied nitrogen" occurred at the Bardowie farm at Hautapu". The quote was obtained out of context because "conservative" was used by Selvarajah *et al*. (1994) meaning "cautious" not "purposely low" as implied in the DRI report.

Apparently the denitrification estimate for the Lichfield site has been made from the ground water data from the Bardowie farm at Hautapu. According to the estimate, denitrification rates at the Hautapu site could be between 600 to 780 kg N/ha/year with a N loading of 1100 kg/ha/year. The basis for this estimate was not provided in the supplementary report by DRI. However, it appears that the estimates have been based on ground water data for the Bardowie farm. It also appears that the approach adopted must have been similar to that reported in the Anchor Products report submitted at the joint hearing for the Buxton farm irrigation project. The denitrification estimate is based on net annual hydraulic loading, throughflow, background ground water nitrate levels and nitrate levels `downstream' of the Bardowie site. The throughflow estimate for Bardowie was between 1600 and 2900 $m^3$/day (average estimate being 2250 $m^3$/day) and it was assumed that the background (`upstream') ground water nitrate level was 12 g/m^3, `downstream' ground water nitrate level was 37 g/m^3 and net annual hydraulic loading was 910 mm. According to this approach the estimated amount of nitrate-N leaching was 525 kg/ha/year. After considering other N transfers (animal product - 75 kg N/ha/year, transfer of N to non-productive areas - 50 kg N/ha/year and ammonia volatilisation - 25 kg N/ha/year) it was estimated that the balance of the applied-N (i.e. 425 kg/ha/year not 525 kg/ha/year as reported) was denitrified. Thus the denitrification loss was 38% of the applied-N. Such estimates can have high errors attached to them due to uncertainties involved with the estimate of throughflow and average nitrate levels in ground water.

It appears that the `downstream' ground water nitrate level has been obtained from a limited number of ground water sampling sites at Bardowie. The information provided by Selvarajah *et al*. (1994) about the Hautapu irrigation project also selected only 8 out of 18 sites currently monitored due to missing data at 7 of the remaining sites and because records from 3 control sites were omitted. The extent of the ground water pollution at the Bardowie site is much higher than reported by Selvarajah *et al*. (1994) if all the sampling sites are considered. Figure 1 shows average nitrate levels at the Bardowie site using (a) 8 long-term records and (b) 15 short-term records. According to this information the average nitrate level (June 1993 to June 1994) found beneath the irrigation site is 46 g/m^3 not 37 g/m^3 as was assumed previously. Using the approach adopted by Anchor Products, the total N leached is 672 kg N/ha/year compared to the previous estimate of 425 kg N/ha/year. Consequently, the estimated denitrification at Hautapu should be 278 kg N/ha/year which is 25% of the applied-N. Note that according to the DRI estimate, the lowest denitrification at the 1100 kg N/ha/year nitrogen loading rate is 600 kg N/ha/year.

It must be emphasised that nitrate leaching is influenced mainly by net hydraulic loading, and the extent of nitrification in soil. Thus nitrate leaching can vary seasonally regardless of N loadings. Using `downstream' ground water nitrate levels for June 1990 to June 1991 (65 g/m^3) the estimated leaching loss is more than 900 kg N/ha/year. Note that the 1990/91 irrigation season received a lower N loading than the previous two seasons.

The important point to note from the above observations is that soil denitrification and nitrate leaching estimates are made with some crude assumptions which subsequently contribute to high errors. The leaching losses obtained from such estimates have to be viewed with caution which can either give rise to apparent high or low leaching losses. Consequently, gaseous N loss estimates made using such leaching values are also highly susceptible to errors. Moreover, nitrogen transformation dynamics are site specific and hence a careful approach is required for estimating potential leaching losses. Information obtained from one location can be used for another, only when the conditions are similar.

The following reasons strongly suggest that the Anchor Products estimate of denitrification loss for the Lichfield site (190 kg N/ha/year) is an overestimate:

1. On a percent loss of applied-N basis the estimated loss of N through denitrification should be 32 kg N/ha/year (8% of 400 kg N). It is well known that most gaseous losses are directly related to the level of their sources applied per unit area.

2. The 8% loss estimated by Russel and Lindsey (1990) could be lower considering the diurnal fluctuation of denitrification losses.

3. Soil pH for the Lichfield irrigation site (5.6) is much lower than that of Bardowie (7.0) and hence there is less denitrification potential at the Lichfield site. Moreover, there could be a high loss of N through ammonia volatilisation from the Bardowie site due to high soil pH. Ammonia volatilised from urea-N or ammoniacal-N present in soil is generally greater when pH is >7.0.

4. Although high rainfall and irrigation can enhance denitrification processes by saturating the soil, similar conditions can also result in a substantial amount of nitrate leaching below the zone of denitrification (i.e. top soil). High rainfall and irrigation can also leach urea-N from urine patches below the same zone,

reducing the denitrification potential. Under such conditions, the rate limiting factor will be the absence of nitrate in the zone of denitrification.

5. Soil types are different for both sites. The Lichfield site has a well drained soil whilst the Bardowie site has a mixture of poorly drained (Te Kowhai silt loam) and well drained (Horotiu sandy loam) soils. The potential for denitrification is much higher for the Te Kowhai silt loam, whilst this soil type also has a low nitrate leaching potential.

It is concluded that it is extremely difficult to predict soil denitrification loss from applied waste water even for a site such as Bardowie where some information is available. Therefore it is even more difficult to predict denitrification for a new site such as Lichfield. While it is acknowledged there will be denitrification from applied waste water after nitrate is formed in soil, the extreme leaching conditions which exist at the proposed site may enhance leaching loss of nitrate. As a cautious estimate a loss of 10% of the applied-N is taken into consideration which means at a loading rate of 400 kg N/ha/year the loss sustained through denitrification will be 40 kg N/ha/year.

(iii) Denitrification due to the presence of animals

It is difficult to separate denitrification losses from irrigated effluent and denitrification losses from urine spots. Generally, irrigation of effluent is carried out soon after a grazing cycle and hence the measurements made by Russel and Lindsey (1991) may have fully or partly included the denitrification losses due to the presence of animals. Sherlock *et al*. (1992) estimated that from intensively grazed New Zealand pasture up to 10 kg N/ha/year of denitrification can occur. As pointed out earlier, the potential for denitrification loss from urine spots can be reduced due to the proposed irrigation practices at Lichfield (leaching below the zone of denitrification) compared with conventional dairy farming practices where little or no irrigation is used.

(iv) Denitrification in ground water

No ground water studies have been undertaken thus far to assess the extent of denitrification in ground water. Moreover, accurate field measurements of denitrification losses are technically difficult to achieve. From available data, a shallow aquifer (4 m depth) with a nitrate concentration of 3.8 g/m^3, and comparable with ground water beneath the Buxton farm denitrified < 0.014 mg N/L/day (Fontes *et al*., 1991). Considering the annual hydraulic loading of 1018 mm at Lichfield, the amount of water available for denitrification treatment would be 10180 m^3/ha. Assuming a denitrification rate of 0.01 mg N/L/day the amount of N denitrified for the irrigation period (270 days) will be 28 kg N/ha/year. This could be a substantial overestimation because the Fontes *et al.* (1991) ground water temperature was 27-31.9^{o}C whilst that of the Lichfield site is about 10-15^{o}C. Since the temperature is directly related to the denitrification rate, it is estimated that only 14 kg N/ha/year could be denitrified. Selvarajah *et al*. (1994) indicated that significant ground water denitrification occurs only in the presence of reduced iron (Fe^{2+}) in ground water (autotrophic denitrification). Such waters have little or no nitrate. As a rule of thumb, Waikato bore waters containing > 0.2 g/m^3 Fe^{2+} have < 1 g/m^3 nitrate-N. The available information for Lichfield suggests that ground water beneath the irrigation site is likely to have little or no reduced iron.

It is concluded that denitrification losses have been overestimated by Anchor Products, and with the available information, my estimation of total denitrification loss at the Lichfield site is up to 64 kg N/ha/year (considering a denitrification loss of 40 kg N/ha/year from soil due to waste water irrigation, 14 kg/ha/year from ground water and 10 kg N/ha/year due to the presence of animals). It must be emphasised that overestimation of the N loss process will lead to excessive

N build-up in soil and the subsequent leaching losses. Considering the complexity of N transformation dynamics, from a resource management point of view it is suitable to use conservative estimates.

3.3.4 Transfer to non-productive areas

Nitrogen transferred from grazed areas onto non-productive areas such as raceways has poor removal mechanisms and hence is susceptible to accumulation and leaching. On the other hand, N transferred to the dairy shed can be applied onto land through irrigation. The proposed irrigation system is intended to spray dairy shed effluent onto the irrigation area from the existing treatment ponds. The amount of N applied through dairy shed effluent irrigation will be approximately 30 kg/ha/year (assuming N excreted in the dairy shed = 20 g/cow/day, number of milking days = 270 days, herd size = 915 cows, and irrigated land area = 164 ha). However, since within 2-3 days about 50% of the N is lost as ammonia from the effluent, the amount of N returned through spray irrigation will be 15 kg/ha/year. Consequently, N loading from dairy shed effluent balances with N loss as ammonia.

3.3.5 Biological nitrogen fixation (BNF) by clover

(i) Effect of high fertiliser-N use on clover-N fixation

As the DRI report indicated, use of *N fertilisers* can reduce symbiotic fixation of nitrogen. The extent of BNF by clover per land area can be affected by

(a) the reduction of clover plant population and

(b) reduction in N fixation through the reduction in rhizobial activity.

A sequence of pasture responses to applied N through *high* fertiliser-N addition can lead to clover plant suppression:

(i) ryegrass responds more rapidly to mineral-N applied than clover, and this response will lead to reduced nutrient and light availability for clover,

(ii) despite the reduction in clover, high fertiliser-N application combined with rapid ryegrass growth will result in high herbage-N,

(iii) high herbage-N content will result in greater ingestion of N by grazing animals,

(iv) high N ingestion by grazing animals will result in greater N levels in urine,

(v) greater ryegrass dominance in urine patches and the cycle continues again from (i).

Ryegrass dominance is easy to assess, and requires only an assessment of the vegetative composition of pasture in question. However, clover number in pasture will not necessarily indicate the extent of BNF. The presence of clover is directly proportional to the amount of BNF in pasture only when the N supplying capacity of a soil is poor or limited (Ledgard *et al*., 1987). When there is sufficient mineral-N present in the rhizosphere, clover derives its plant N from soil mineral-N rather than through symbiotic N fixation. Under these conditions the number of nodules per clover plant declines or the nodules become inactive. A decline in root nodules is often the first, and most noticeable symptom that clover plant N is derived from soil mineral-N.

(ii) Effect of high waste water application on clover-N fixation

There is little information available on the effect of waste water application on the extent of BNF by clover. However, it is emphasised that using clover suppression data from fertiliser-N studies to estimate BNF in pasture applied with high-N waste water at high hydraulic loadings can only result in misleading estimates of clover-N fixation due to following reasons:

(i) In freely draining soils, high frequency of waste water application will lead to removal of mineral-N (nitrate-N) and urine-N through leaching. Although leaching occurs in a non-irrigated dairy pasture system through rainfall, the extent of leaching is much greater when pasture is irrigated at 50-75 mm every 15 days.

(ii) In a non-irrigated pasture system, clover performance is often affected by soil moisture deficit (Ledgard *et al*., 1987). Soil moisture is not limited when pasture receives frequent irrigation.

(iii) A high proportion of the dairy factory waste water contains organic-N. Organic-N generally enhances clover performance (e.g. the clover dominance in dung patches (Weeda, 1967)). Although the DRI report argues that protein breakdown is rapid in soils, when compared with mineral-N availability from fertiliser-N the mineralisation of protein is a slow process. Protein is insoluble in water and when it is decomposed it goes through aminization followed by ammonification reactions to yield ammonium. In contrast, fertiliser such as urea is readily soluble in water and yields ammonium within a few hours of application through hydrolysis.

(iv) High fertiliser-N use can reduce soil-C in many ways (soil-C depletion) and soils with low available-C are not conducive for heterotrophic bacterial activity. Note that rhizobium also depends on soluble-C for dissimilatory reduction of nitrate-N and hence the presence of soluble-C can enhance rhizobial activity. Increased rhizobial activity in turn will increase bacterial inoculation of clover roots.

(v) High use of fertiliser-N can lead to soil acidity which will suppress clover fixation of N. In the case of dairy factory waste water application, soil is not acidified despite the high N loading due to inherent waste water alkalinity.

According to the supplementary information provided by Anchor Products about 30% of the pasture at the Lichfield site consists of clover which corresponds to the vegetative composition of typical dairy pasture in the region. Studies performed in the Waikato region by AgResearch indicate that BNF in well established pasture is up to 280 kg N/ha/year (Ledgard *et al*., 1990). As a rule of thumb, N input to a grazed pasture in the Waikato region could be 200 kg N/ha/year. The question is whether a similar amount will be contributed in a grazed pasture system receiving dairy factory waste water at high N and hydraulic loadings. Visual assessments made during summer 1993 and autumn 1994 at the Hautapu dairy factory waste water irrigation site suggest that clover is abundant. Samples of clover showed that there was a substantial amount of nodulation and the nodules were active (presence of `leghaemoglobin' in nodules).

The only available information on clover behaviour under continuous organic waste water irrigation (17 mm/week) suggests that clover is capable of fixing up to 241 kg N/ha/year under Waikato conditions. The waste water used was dairy shed effluent with 74% organic-N. The result was obtained using an indirect BNF assessment from a lysimeter study conducted on a Te Kowhai silt loam with a controlled water table (Barkle *et al*., 1993).

According to the DRI estimate, clover fixation of N at the Lichfield irrigation site will be 0-60 kg N/ha/year. Such an estimate assumes that there would be zero BNF under certain conditions. It must be emphasised that for zero BNF to occur under irrigated conditions, there should be little or no clover present in the pasture. If the waste water irrigated pasture contains a substantial amount of clover, which has active root nodules as at the Hautapu site, the BNF should be about 200 kg N/ha/year. This estimate is equivalent to the average BNF for well established pasture for the Waikato region (Selvarajah, 1994). This estimate considers traditional white clover cultivars such as *Huia* (a greater amount of BNF can occur in other cultivars such as *Aran, Kopu or Pitau*). It is assumed here that the traditional white clover *Huia* is still in use at the Lichfield site.

It must be noted that at the Hautapu (Bardowie) farm site, clover appears to disappear and reappear during certain seasons (J. Barnett, DRI, pers. comm.). This is probably due to saturated conditions caused during wet weather conditions combined with high hydraulic and sodium loading rates. Trials indicate that white clover (*Huia*) is able to fix about 82 kg N/ha/year in the first year following planting (Ledgard *et al*., 1990). This amount could be halved (i.e. 40 kg N/ha/year) for clover receiving waste water irrigation due to deleterious effects during wet weather conditions. Due to relatively high hydraulic loading combined with the sodium loading the BNF at Lichfield could be approximately 40 kg N/ha/year.

3.3.6 Leaching losses under conventional dairy pasture systems

Most Waikato soils have a high nitrifying capacity (oxidation of ammonium to nitrate) due in part to their volcanic origin. This high nitrifying capacity emphasises the requirement for stringent hydraulic management for these soils; mismanagement can lead to severe nitrate leaching losses which will result in ground water nitrate elevation. There have been several studies conducted on the extent of leaching under various farming practices. The majority of these studies have used lysimeters for the estimation of leaching losses. Actual field studies should examine the ground water to obtain an accurate assessment of the extent of leaching. One such estimate performed in the Hamilton Basin area showed that under ample rainfall conditions the leaching of nitrate from farmlands could be up to 60 kg N/ha/year (Selvarajah *et al*., 1994). The study area is used predominantly for dairying and cropping (especially maize). Cropping involves cultivation of soil and this could leach a substantial amount of nitrate. Thus the above leaching estimate (60 kg N/ha/year) could be considered as an overestimate for conventional dairy farming in the Waikato region (clover-ryegrass pasture with little or no input of fertiliser-N and high input of phosphorus).

Table 3. Nitrogen balance

Nitrogen Transformation Processes	**Anchor Products Estimate**	**Staff Assessment**	**Recommended**
	Input 400 kg N/ha/year		**Input 300 kg N/ha/year**
Products (milk + maintenance)	75	75	85
Ammonia volatilisation	25	25	25
Denitrification	190	65	55
Clover-N fixation	+60	+40	+40
Leaching loss under clover based pasture systems	-	60	60
Non leaching N losses Nitrogen available for leaching	290 170	165 275	165 175

According to Table 3 it is clear that for the proposed application rate of 400 kg N/ha/year my estimate for net leaching will be 275 kg N/ha/year. I emphasise that Anchor Products has overestimated the denitrification loss as 190 which resulted in an underestimation of nitrate leaching loss. Considering the difficulty in predicting such complex and dynamic N transformation processes in soil and water for a specific site, such as Lichfield, and the poor information available on the potential fate of leached nitrate, it is environmentally safer to adopt conservative N loading rates. Ideally the applicant should have performed some work on N transformation potentials for the site without speculating the N loss processes. Thus it is concluded that the proposed rate of N application by Anchor Products is likely to cause excessive nitrate leaching and ground water contamination. From the available information, 300 kg N/ha/year application rate appears to have considerably less leaching (at least 100 kg N/ha/year) than that of the proposed application rate.

3.4 Farm nitrogen management strategies

Farm management should be aware of the high annual N application rates through waste water irrigation and hence take account of other N inputs. Paddocks which receive waste water from Anchor Products should not be used for other forms of N application. This includes fertiliser-N application, receiving and using N based waste water from other waste water sources (except for Anchor Products) and using stock feed (e.g. silage, hay or concentrates) brought from other farms or suppliers unless there is an acute shortage of stock feed.

Another vital N management practice is control over cultivation practices. Since ploughing during warm periods is considered to be a major source of ground water nitrate (Francis *et al.*, 1993), any form of cultivation should not be performed on the irrigated land. If ploughing is required for the reestablishment of the grass cover or maize cropping, it should be performed during dry weather in June or July to minimise mineralisation from soil organic-N and subsequent leaching of nitrate. Since irrigation is not performed during these months this period provides the best time for any cultivation.

4. EFFECTS OF PHOSPHORUS AND SODIUM

Soil test results for the Bardowie farm clearly show that there is an excessive amount of phosphorus (P) and sodium (Na) present and an optimum level of calcium (Ca), magnesium (Mg) and potassium (K). In the case of soil build-up of Ca, P, K and Mg there is generally no adverse effect on soil fertility, plant and animal health or ground water quality. Several years of P build-up in soil can eventually cause P leaching. However, pumice soils have one of the highest P retention capacities and hence leaching of phosphorus will be unlikely at the Lichfield site.

According to the proposed waste water volume application (3050 m^3/day) the annual loading rate of Na at the Lichfield site will be 205 tonnes. On an area basis this will be approximately 1250 kg Na/ha/year in the spray area. High Na levels in soil will destroy soil structure which is essential for soil aeration and drainage. When clay minerals are dispersed by Na they clog soil pores causing poor drainage. The success of the proposed irrigation system depends mainly on good soil drainage conditions. Although yellow brown loams are less affected by high soil Na levels than many other New Zealand soils, caution must be taken in applying such large quantities of Na. The mass loading of Na can outweigh any benefit that is obtained by having other cations such as K, Mg and Ca. It must also be noted that Na can be more deleterious to soil structure when applied with waste water which has a pH > 9.0. Due to presence of high Na levels it is important to keep the waste water pH below 8.5 throughout the irrigation period. As a soil amendment practice, application of high amounts of calcium can leach excessive Na from soil. Lime ($CaCO_3$) is a cheap source of calcium and can be used to leach Na. However, since lime can also cause elevation of soil pH, it is discouraged as a source of calcium. Gypsum ($CaSO_4$) is the most suitable amendment chemical under the given conditions and will not elevate soil pH and supplies additional sulphur for pasture growth. It is recommended that gypsum should be applied annually to avoid any deleterious effects caused by Na and excessive accumulation of Na.

It should be noted that the results from the Hautapu irrigation site indicate that despite the resilient nature of allophanic soil such as Horotiu sandy loam to flocculation caused by high sodium use, the bulk density of this soil has increased substantially. Although the use of lime has reduced increases in the bulk density of surface soil (0-75 mm depth), the bulk density of the balance of the soil profile remains much greater than determined in 1984. For example, the bulk density at soil depth 75-150 mm measured in 1984 was 815 kg/m^3 whilst, the 1992 measurement was 1180 kg/m^3. Increasing bulk density indicates compaction of soil and poor drainage. Although the proposed amount of sodium use at the Lichfield site (1250 kg Na/ha/year) is much less than that of the Hautapu site (2000 kg Na/ha/year), sodium should be used with caution on any silt soil such as the Taupo silt loam due to rapid flocculation of silt. Such soils have low resilience to applied sodium and such application can result in compaction and overland runoff.

The major Na based chemical that will be used at the Lichfield cheese factory is caustic soda (NaOH). It is recommended that alternative non-Na based chemicals should be used as much as possible. For example, potassium hydroxide (KOH) is a good substitute for NaOH. Every effort should be taken to recycle the use of NaOH without discharging into the waste water.

The extremely variable pH levels of waste water applied can cause some long-term soil problems along with the problem caused by Na- pH interaction. Bardowie farm soil pH levels have increased by a unit within a decade of waste water application (from 6.2 (1982) to 7.1 (1992)). At pH 7.0 certain micronutrients (e.g. copper, selenium) that are essential for plant and animal health will be deficient. This is a farm management problem and can be solved by providing a supplementary diet containing trace elements. Most soil biological processes including nitrification (the biological oxidation process of ammonium to nitrate form) is best at neutral pH (7.0) and hence increased soil pH can result in greater nitrate production in soil. On the other hand, high soil pH can result in a greater proportion of nitrogen gas being released (N_2) during the denitrification process. Acidic soils tend to produce more nitrous oxide gas (N_2O) which is considered as an ozone layer depleting and green house gas. However, since the extent of denitrification loss is substantially less than leaching loss of nitrate, increased pH can have greater environmental effects on the receiving environment than maintaining the existing level (5.6).

5. ADVERSE EFFECTS DUE TO MISMANAGEMENT OF WASTE WATER TREATMENT, USE OF CERTAIN CHEMICALS AND ADVERSE WEATHER CONDITIONS

The proposed irrigation project will cause some adverse effects on the environment under the following circumstances:

5.1 Soil

1) There will be sodium accumulation in soil due to the presence of sodium hydroxide in the waste water. Sodium should be `flushed' out by using either lime or gypsum regularly. Otherwise sodium accumulation will cause clay/silt flocculation which will increase bulk density. The problem is exacerbated due to the presence of grazing animals. This will result in soil compaction and poor drainage conditions and subsequently contribute to surface runoff.

2) Soil pH will elevate due to the application of waste water containing sodium hydroxide and the subsequent use of lime to `flush' sodium out. Elevation of soil pH will result in poor availability of certain essential micronutrients for plants and animals. Both lime and sodium hydroxide are alkaline materials. The use of gypsum, however, will help to maintain the background pH. Until extensive research is performed, it is not known about the effects of elevated soil pH on biodiversity.

3) Considering the presence of welded ignimbrite rock beneath the `shallow' soil profile excessive application of waste water may result in saturated soil conditions due to water perching.

5.2 Ground water

1) Frequent ripping of soil will result in preferential flow paths and poor treatment of waste water. Presence of preferential flow paths will also lead to excessive urea-N leaching from urine voided by grazing animals. Excessive ripping during warm seasons will cause high mineralisation of organic-N and will result in nitrate leaching.

2) Paddocks receiving greater N loading rates or hydraulic loading rates will result in greater nitrate leaching.

3) Excessive nitrate elevation will occur (i.e. > 10 g nitrate-N/m^3) in shallow aquifers (immediate receiving environment whose presence is yet ambiguous) which have low throughflows, aquifer thicknesses and storativities.

4) There will be seasonal elevation of ground water nitrate levels due mainly to high hydraulic loading combined with warm weather conditions.

5.3 Surface water

1) Occasional heavy storms are characteristics of the irrigation site. Such heavy storms cause surface runoff in the catchment. Waste water applied onto land during heavy rainfall will result in surface runoff and may exit the site and pollute surface water.

2) Excessive leaching of nitrate to ground water may result in surface water pollution of nitrate through subsurface flow.

6. RECOMMENDATIONS

6.1 Hydraulic loading

Objective: *Maintaining a hydraulic loading which does not enhance nutrient leaching, surface ponding and surface runoff.*

(a) Every effort should be taken to minimise waste water generation in the factory.

(b) The hydraulic loading should not exceed 50 mm per application.

(c) The irrigators should be able to deliver the waste water at a range of application rates suitable for given weather conditions, i.e. 2-6 mm/hour.

(d) Considering the high N, BOD and hydraulic loadings, a 15 day irrigation cycle should be used.

(e) No surface ponding or surface runoff should be apparent during the entire irrigation operation.

6.2 Nitrogen loading

Objective: *Avoid or minimise soil and water contamination.*

(a) Total nitrogen loading should not exceed 300 kg N/ha/year.

(b) The total land area available for irrigation should not be less than 218 ha. Alternatively, an area of 164 ha can be used by reducing N in the waste water (e.g. annual N loading of 49200 kg).

(c) Fertiliser N use or other N based waste water should not be applied in areas which receive 300 kg N/ha/year through waste water irrigation.

(d) Stock should not be fed with stock feed from external sources (e.g. silage, hay, concentrates) other than the pasture from the irrigated area which receives 300 kg N/ha/year waste water irrigation.

(e) Cultivation of irrigated areas should be permitted only during dry weather in June or July.

(f) Environment Waikato should be informed if new chemicals are introduced into the waste water.

6.3 Sodium loading

Objectives:
(a) Avoid or minimise the use of sodium hydroxide; and
(b) Reduce sodium in the waste water and limit sodium elevation in soil water.

(a) The proposed Na loading rate should be reduced either through the use of substitute chemicals (e.g KOH for NaOH) or more efficient recycling of waste water.

(b) An appropriate amount of gypsum should be applied annually.

7. ENVIRONMENTAL MONITORING (COMPLIANCE AND EFFECTS)

7.1 Objectives of monitoring

The irrigation system is unlikely to cause any adverse environmental effects provided the recommended N loading rate is used and a proper waste water management (application and control of chemicals) is carried out. However, land treatment systems are site specific (influenced mainly by soil type, pasture performance, landscape, climate). Moreover, the proposed waste water irrigation involves high variation in effluent loading (volume) and highly variable waste water quality, even on a daily basis. Seasonal effects on nutrient dynamics and ground water level changes can also affect the amount of pollutants present in receiving waters. Any irrigation system is susceptible to accidents and a record documenting all accidents should be maintained (e.g. excessive irrigation or spillage). All these factors emphasise the requirement for systematic monitoring to ensure that the predicted environmental effects are not exceeded. Furthermore, compliance monitoring is required to ensure the system is managed according to the consent conditions.

As indicated before due to the lack of information available on the receiving environment for the Lichfield site it is virtually impossible to design and construct a specific monitoring programme for environmental impact monitoring. Consequently, the monitoring programme outlined here will indicate the type of monitoring and number and frequency of sampling requirements rather than the locations of sampling points.

7.2 Hydraulic loading

Objective: *To monitor the amount and effect of hydraulic loading*.

(a) A daily record of irrigated waste water flow and area irrigated should be kept.

(b) A daily record of rainfall should be kept.

(c) Overland runoff (if any) and its destiny should be monitored whenever this occurs.

7.3 Waste water quality

Objective: *To monitor loading rates of nutrients.*

(a) The irrigation water should be characterised for the following properties on composite samples:
(a) daily- pH, BOD, TKN and nitrate-N

(b) monthly- Na, Ca, Mg, K and TP.

7.4 Ground water quality

Objectives: *To monitor (a) the annual and seasonal effects of waste water irrigation on ground water quality and (b) the effects of waste water treatment on ground water quality.*

(a) At least 10 sampling sites are required for monitoring nitrate in leachate at 1000 mm depth. These are basic soil solution samplers (e.g suction cups) which should contain a proper preservative and should be sampled monthly for nitrate-N and nitrite-N. These samplers should be distributed evenly on hill and valley soils.

(b) At least 25 sampling sites are required for ground water quality monitoring for perched ground water (this still requires confirmation), and Whakamaru and Waiotapu aquifers. These sites may include existing bores and piezometers on the site.

(b) Piezometer and/or bore water samples should be monitored on a monthly basis for ground water level, pH, conductivity, nitrate-N, TKN, nitrite-N, ammoniacal-N, sodium, total-P and dissolved organic carbon.

7.5 Surface water monitoring

Objective: *To monitor the annual and seasonal effects (mainly ground water contribution to nutrient levels in surface water) of the waste water irrigation onto land.*

(a) Proper sampling sites (possibly using sites monitored previously) should be established with the assistance of EW technical staff for upstream and downstream samplings. The Pokaiwhenua and Ngutuwera streams should be monitored for possible contamination through subsurface flow.

(b) Surface water should be monitored for the following on a monthly basis: flow rate, pH, temperature, conductivity, nitrate-N, TKN, ammoniacal-N, total-P and BOD.

7.6 Soil health

Objective: *To monitor the annual effects of waste water irrigation on soil quality.*

From the sustainability view point, I am not certain about the long-term effect of increasing soil pH on biodiversity. Diversity is often a measurement of uniformity and/or degree of dominance or heterogeneity of a community. Diversity indices have been widely used to measure the effects of certain activities on these communities. It is generally believed that species diversity is a measure of order within a community. Biodiversity measurement is a complex and costly process. Having measured, it is very difficult to interpret due to poor information availability for comparison purposes. However, the use of existing simple indices (e.g. measurement of enzymic activity such as sulphatase or urease) can reasonably indicate the degree of biological and chemical degradation caused by certain activities. Thus monitoring such indices should be part of the effects monitoring programme for the proposed irrigation project.

(a) Prior to the irrigation, and thereafter every 3 years during the duration of any discharge permit, the soil should be tested for clay mineralogy, biomass activity, enzyme activity (e.g. invertase, urease, sulphatase, phosphotase), and potentials for denitrification, nitrification, and N mineralisation. Samples should be obtained from representative hill and valley soils from 0-1000 mm at 100 mm increments.

(b) Soil should be monitored for the following characteristics annually at the end of April: infiltration rates, bulk density, pH, TKN, ammoniacal-N, nitrate-N, organic-C, calcium, sulphate, total-P, sodium, potassium and magnesium. Three sampling sites for hill soils and 3 sampling sites for valley soils should be obtained. Each sampling site should be a composite of 3 sampling depths, 0-200, 200-600 and 600-1000 mm.

7.7 Data requirements

Sample collection, storage, analysis and reporting of results shall be carried out to an approved standard. Sample results shall be forwarded to Environment Waikato within 6 weeks of the commencement of each month being reported upon. Results shall be provided in an electronic and hard copy format.

8. SUMMARY

According to the staff report the nitrogen loading rate for the proposed irrigation system is set at 300 kg N/ha/year. It has been identified that even at this rate of application there will be leaching loss of nitrate and subsequent ground water contamination. However, these effects are likely to be minor. Thus the rate is considered to be environmentally and agronomically sustainable. According to the recommended annual N loading rate the land area available for waste water irrigation should be increased from 164 ha to 218 ha.

The N loading proposed by Anchor Products (400 kg N/ha/year) is considered to be excessive for the system proposed. The proposal's recommendation was based heavily on anticipated high denitrification losses. The staff report illustrates that such high denitrification is not possible for the proposed system, and hence involves high environmental risk.

It is stressed that the information provided on the receiving environment is very poor and hence it is difficult to predict the environmental effects and recommend monitoring sites for ground and surface water sampling. The nitrogen loading rate recommended by this report is still considered to be environmentally sustainable despite the lack of information on the receiving environment.

A strong recommendation has been made for reducing sodium in the waste water stream. It has been identified that high sodium levels in soil will cause soil compaction. Considering the rolling landscape of the site this issue has to be considered seriously due to potential surface runoff of applied waste water.

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