# Treatment of farm effluents in New Zealand's dairy operations

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## ABSTRACT

Maintaining growth through intensification in the New Zealand dairy industry is a challenge for various reasons, in particular sustainably managing the large volumes of carbon and nutrient rich effluent. Dairy farm effluents have traditionally been treated using pond systems prior to waterway, and more recently land application. Ponds are effective in the removal of carbon and suspended solids, however they are limited in their ability to remove nutrients. Current environmental concerns associated with the direct discharge of effluents to surface waters have prompted the development of technologies to either minimise the nutrient content of the effluent or apply effluents to land. This paper discusses various approaches and methods for the treatment of effluent to enable dairy farmers to sustainably manage farm effluents. This includes advanced pond treatment systems, stripping techniques to reduce nutrient concentration, land application strategies involving nutrient budgeting models to minimise environmental degradation and enhance fodder quality, and deferred irrigation.

## INTRODUCTION

In many countries, dairy farm effluents are treated biologically using pond based systems. This treatment often removes much of the biological oxygen demand (BOD) and the suspended solids of the waste. Pond systems, however, are not primarily designed to remove nutrients, such as N, P and K. Nutrients remaining in farm pond effluents can be significant pollutants and when discharged to streams stimulate weed and algal growth, and result in the eutrophication of the waterways (Houlbrooke et al., 2004a; Wang et al., 2004). In New Zealand, with the introduction of the Resource Management Act (1991), discharge of effluents to surface waters is now a controlled or a discretionary activity that requires resource consent (Selvarajah, 1999; Wang et al., 2004). Commonly, resource consent approval will require effluent nutrient concentrations to be minimised before entering surface waters. This can be achieved by nutrient stripping of effluents via advanced secondary and tertiary treatment, or through land application. In New Zealand, many Regional Councils encourage land application of farm effluents (Houlbrooke et al., 2004a). If designed and managed appropriately, returning dairy shed effluent directly to land will invariably minimise the impact of effluent discharge on receiving aquatic environment while also providing a valuable source of water, nutrients and carbon to soils. In many instances this may also be the cheapest and most socially/culturally accepted form of final treatment (Wang et al., 2004).

Optimum use of effluent and manure by-products requires knowledge of their composition and treatment processes, not only to maximise their benefits, but also to minimise environmental damage (Houlbrooke et al., 2004a). Environmental concerns associated with the land application of effluent and manure by-products from confined animal industries encompass all aspects of non-point source pollution, including contamination of surface water with soluble and particulate P, leaching losses of N in subsurface drainage to groundwater, movement of microbial contaminants, reduced air quality by emission of volatile organic compounds, and increased metal input to soils (Bolan et al., 2004c; Bhandral et al., 2007; Houlbrooke et al., 2008). Maintaining the quality of the environment therefore must be a major consideration when developing management practices to effectively use effluent and manure by-products as a nutrient resource and soil conditioner in agricultural production systems (Sharpley et al., 1998).

Given the potential value of farm effluents, increasing research resources are now being committed internationally to develop improved systems able to convert effluent and manure based wastes to a valuable and environmentally safe resource. This paper provides an overview of recent changes in New Zealand dairy sector, the volume and treatment of effluent produced in dairy operations and the integrated management of these effluents into farming practice in relation to sustainable production and environmental protection.

## RECENT CHANGES IN DAIRY SECTOR

There have been major changes within the New Zealand dairy industry over the past fifteen years, with growth as a sector increasing steadily over this time. Concurrently the contribution to agricultural gross revenue and agricultural exports has also expanded, resulting in large areas of pastoral land (especially sheep and beef farms) being converted to dairy farms. In the past twenty years, the number of dairy farms has fallen by 21% (LIC, 2008), yet average farm and herd sizes during this time have increased substantially. This trend is most prevalent in the South Island of New Zealand where the move from small single-operator farms to larger more complex syndicate-owned farming enterprises is more evident (Table 1). Although ‘payout prices’, $NZ dollar per kilogram of milk solids, have remained relatively stable, herd production and farm cost efficiencies have improved substantially (Table 1) (LIC, 2008).

**Table 1.**Major changes in New Zealand dairy industry (LIC, 2008).

| Year | | 1950 | 1977 | 2006/07 |
|---|---|---|---|---|
| No of herds | | 34367 | 17363 | 11630 |
| | North Is. | | | 9,343 |
| | South Is. | | | 2,287 |
| Cows/farm | | 54 | 112 | 337 |
| | North Is. | | | 296 |
| | South Is. | | | 505 |
| Farm size (ha) | | | 56 | 121 |
| | North Is. | | | 107 |
| | South Is. | | | 179 |
| Milk (L/cow) | | 2387 | 2787 | 3791 |
| Milk solid (kg/cow) | | 191 | 223 | 330 |
| Milk solid (kg/ha) | | | 653 (1992) | 934 |
| No of cows (million) | | 1.82 | 2.08 | 3.92 |
| No of major Co-op (NZ) | | 231 | 116 | 3 |

## QUANTITIES OF DAIRY EFFLUENT PRODUCED

In countries such as Australia and New Zealand where open grazing is practiced, large amounts of animal excreta including dung and urine are deposited directly onto the pasture.

On dairy farms however, approximately 6- 10% of excreta is deposited in the milking shed and collecting yards. When the yards and milking area are cleaned with high-pressure hoses, farm dairy effluent is generated at approximately 50 L per cow per day (Mason, 1997; Selvarajah, 1999). While this estimate holds good for most modest sized farms, there is nevertheless a significant variation. A recent survey suggests that larger scale farms with herds greater than 3,000 cows can use 25 to 31 L/cow/d (Hill and Lowe, 2009). External runoff from roof, milking shed yard, adjacent feed pads and stock races contribute to the effluent collection although current design features aim to minimize or exclude these.

It is estimated that annually in New Zealand about 70 million $m^3$ of effluent are being generated from dairy sheds (Saggar et al., 2004). The effluent contains significant quantities of valuable nutrients that could be applied onto land in order to improve soil fertility and increase the sustainability of farming systems. Bolan et al. (2004b) estimated the value of effluents from dairy shed and piggery farms in New Zealand to be in the order of $21 million per year.

With intensification of the New Zealand dairy industry underway, more and more farms are temporarily removing stock from pastures, either to protect soils from compaction and pugging damage, or to more adequately provide feed at times of the year when pasture growth does not meet demand. This has given rise to specialised facilities now being built on New Zealand dairy farms, such as stand-off pads, feed pads, loafing pads and sheds (Luo et al., 2006). Where reducing soil damage is important, stand-off pads are being constructed as areas to hold stock for up to 18 hours a day. Where increasing feed intake or utilising feed supplements more efficiently is an issue, feed pads are used. Some systems combine both the standing-off and feeding into one facility like a self-feeding pad (sometimes referred to as an out-wintering system). Going one step further where climatic conditions are more severe, covered systems such as wintering barns and herd homes are now more common (Longhurst et al., 2006).

Due to the partial or total confinement of cows during the wet winter period, large quantities of effluent and manure are produced (Longhurst and Luo, 2007). The characterisation of manure will vary depending on its source, animal feeding regime and how it is collected, stored and treated (Table 2). The volume and concentration of stock excreta on any wintering pad will be influenced by the intensity of use and type of feed provided. In addition to stock excreta the volume of effluent may be increased through cleaning or rainwater. Where for one reason or another there is a high rate of shed water used, a comparatively large volume of low density effluent is produced. Conversely, low rates of shed water use lead to smaller volumes of effluent with a more concentrated consistency. This range of volumes and concentrations has its attendant range of logistical and consent compliance implications.

Typically, effluent from stand-off or feed pads adjacent to milking sheds is collected via the farm dairy shed effluent system. On stand-off/feed pads, which are often away from the milking shed, cow manure generally remains within the top 5 cm of the surface material while the urine drains through the pad profile for drainage collection, or in some cases drainage to groundwater. To maintain the pads in an optimum condition for cow health and comfort, the top layer of manure is scraped off regularly and stockpiled. Similar for stand-off/feed pads, solids in the wintering barns are generally removed and stockpiled until soil conditions are suitable for land application (Longhurst and Luo, 2007). In herd homes, animal excreta are stored in the under-floor manure bunkers for several months. After winter use of the herd home, manure volumes in the manure bunkers tend to decline as a result of natural decomposition processes, while the dry matter content increases with time. Nutrient enrichment of the final manure for land application may also increase over time due to evaporative moisture loss (Longhurst et al., 2006).

**Table 2.** Typical nutrient concentrations (%) in various effluents compared to farm dairy effluent (Longhurst and Luo, 2007).

| Source | DM | N | P | K |
|---|---|---|---|---|
| Farm dairy effluent | 0.8 | 0.45 | 0.006 | 0.035 |
| Feed pad | | | | |
| Slurry | 4.0 | 0.150 | 0.030 | 0.100 |
| Effluent post separation | 0.3 | 0.025 | 0.003 | 0.030 |
| Separated solids | 20 | 0.45 | 0.08 | 0.200 |
| Stand-off pad | | | | |
| Solids | 25 | 0.20 | 0.15 | 0.200 |
| Self-feed pad | | | | |
| Scraped manure | 15 | 0.20 | 0.03 | 0.075 |
| Herd Home | | | | |
| Bunker manure | 18 | 0.50 | 0.20 | 0.750 |

It is important to note that typical nutrient concentrations, like typical effluent volumes, are averages within a range. Laboratory analyses of nitrogen in farm dairy effluent from one farm at different times showed 223 and 724 g N $m^{-3}$ respectively, ranging from half to nearly twice the "typical" figure of 450 g N $m^{-3}$ (Hill and Lowe, 2009). Similarly, four laboratory analyses of liquid effluent from a solids separator on another farm, taken a few days apart, showed nitrogen concentrations of 710, 580, 520, and 490 g N $m^{-3}$ respectively (Hill and Lowe, 2009).

## TREATMENT OF DAIRY EFFLUENT USING POND SYSTEMS

### Passive two pond system

In New Zealand and Australia, some dairy farms treat their effluent biologically using two-pond systems (Craggs et al., 2008), where the first pond is anaerobic and the second pond, often termed aerobic, is usually a facultative one, with an aerobic top layer over an anaerobic base. The dual pond system is effective in the removal of suspended solids and carbon (i.e., BOD) from the liquid effluent, however there has been some debate about its efficiency in the removal of nutrients (Mason, 1997). Although some P removal does occur, associated with settling of solid material, and N removal (25-50% of inorganic N) via ammonia volatilisation (Sukias and Tanner, 2005), the large amount of nutrients remaining in the final effluent are considered pollutants when the farm wastes are discharged to waterways (Sharpley et al., 1998).

Porous materials can be used to adsorb nutrients from effluents. For example, zeolite, a naturally occurring, porous and electrically charged alumino-silicate mineral is found to be effective in retaining cations and anions from wastewater (Nguyen and Tanner, 1998). Similarly, organic material such as bark has been found to be effective in the retention of nutrients (Lens et al., 1994). In some cases contaminant-stripped effluents can then be discharged safely into waterways and the nutrient-enriched porous materials can be recycled as a soil conditioner or nutrient source.

A study was conducted at Massey University monitoring the concentration of BOD, chemical oxygen demand (COD), total suspended solid (TSS) and various nutrients in a two pond system over a period of 12 months (Bolan et al., 2004b). The study also examined the role of a bark filter system in removing nutrients during the pond treatment systems. The data on TSS, COD, BOD and nutrients in the effluent samples collected from the anaerobic inlet pipe (i.e., untreated effluent) and the oxidation pond outlet pipe (i.e., treated effluent) indicate that the pond treatment system achieved considerable reduction in the concentration of the first three components (57.1 – 74.3%). There was however, no significant difference in the concentration of nutrients between the anaerobic pond inlet and the aerobic pond outlet. In this particular case most of the N in the effluent was in ammonium ($NH_4^+$-N) form and most P was present as dissolved reactive orthophosphate (DRP). There was a small reduction in the $NH_4^+$-N (8.3%) concentration in the oxidation pond outlet which was attributed mainly to ammonia volatilization and/or microbial immobilization. The data indicated that the two-pond system was effective in removing suspended solids, COD and BOD, but not nutrient ions, other than that associated with organic sediment settling in the anaerobic pond.

Most farm effluents are expected to contain high levels of nutrients, even after biological treatment using pond systems (Hickey et al., 1989; Longhurst et al., 2000). Several factors appear to contribute to the poor nitrification of $NH_4^+$-N seen in the oxidation pond; including high light attenuation thereby creating a shallow euphotic zone where algae can sustain photosynthesis (mean depth only 11 cm) (Sukias et al., 2001) and a consequent shallow oxic zone near the surface that may be limited by the retention and abundance of nitrifying bacteria (Sukias et al., 2003). Contributing to this largely anoxic volume of pond water is the build-up of sludge in the base of the pond, which is likely to reduce the oxygen content and thereby decrease the rate of oxidation of $NH_4^+$-N to nitrate ($NO_3^-$-N) (Mason, 1997). One of the major pathways of N removal in effluent ponds is biological denitrification, leading to the emission of nitrous oxide and di-nitrogen gas (Lowrance and Hubbard, 2001; Groffman and Crawford, 2003). However, unless the $NH_4^+$-N is oxidized to $NO_3^-$-N, subsequent denitrification of $NO_3^-$-N cannot occur. Regular removal of sludge from the oxidation pond can improve the overall conversion of $NH_4^+$-N to $NO_3^-$-N (nitrification) by providing greater opportunities for oxidisation. Sludge is also rich in carbon which is likely to consume oxygen; so removal of sludge will certainly improve nitrification.

Nitrification can be enhanced by providing supplementary aeration in the second pond. Additional benefit is gained also by providing surfaces onto which slow growing nitrifying bacteria can attach and thereby increase the depth of the oxic zone of the pond (Sukias et al., 2003). Where continuous mechanical aeration is used in dairy ponds conversion of $NH_4^+$-N to $NO_3^-$-N may increase by as much as 95-99%. This enhanced nitrification in the treatment pond has implications to the subsequent management of the effluent, including a higher potential for leaching of $NO_3^-$-N and green house gas emission during land application of the treated effluent. Bhandral et al. (2007) for example, noted that the $NO_3^-$-N leaching and nitrous oxide emission during the land application of farm effluents increased with increasing $NO_3^-$-N concentration in the effluent.

Removal of P in pond effluent is enhanced through chemical precipitation using calcium, iron, aluminium or magnesium compounds (Weaver and Ritchie, 1994). Most of the K however remains in ionic form in solution, due to its low demand in microbial growth (Alexander, 1977).

Results from a pilot-scale field experiment conducted in New Zealand (Bolan et al., 2004b) show that *Pinus radiata* bark is effective in the removal of nutrients from dairy shed effluent in a two pond treatment system. The mean concentrations of N in: the untreated effluent, the treated effluent in the absence of bark, and, the treated effluent in the presence of bark were 145.4, 95.4 and 18.7 g m$^{-3}$, respectively.

**Table 3.** Concentration of biological oxygen demand (BOD), chemical oxygen demand (COD), total suspended solids (TSS) and nutrients (N, P and K) after 24 weeks of bark treatment (Bolan et al., 2004b).

| Characteristics | Concentration (g m$^{-3}$) | | | Percent removal | |
|---|---|---|---|---|---|
| | Anaerobic inlet | Aerobic outlet | | | |
| | | Without bark | With bark | Without bark | With bark |
| BOD | 195 | 105 | 55.1 | 46.2 | 71.4 |
| COD | 780 | 395 | 185 | 49.4 | 76.2 |
| TSS | 350 | 105 | 25.5 | 70.0 | 92.8 |
| N | 165 | 110 | 18.7 | 33.3 | 88.7 |
| P | 28 | 24 | 4.8 | 14.3 | 82.8 |
| K | 175 | 168 | 92 | 4.0 | 47.2 |

The N concentrations in the bark treated effluent reached close to the recommended maximum permissible level (MPL) value (TN 0.6 g m$^{-3}$, $NH_4^+$-N 0.2 g m$^{-3}$) in the receiving surface water (Hickey and Vickers, 1994; ANZECC, 2000). It is to be pointed out that the MPL values depend on the dilution caused by the flow in the receiving waters. Although the bark treatment achieved a significant reduction in the concentration of P in the effluent, the concentration was still higher than the recommended MPL value for the prevention of eutrophication (TP 0.03 g m$^{-3}$, DRP 0.01 g m$^{-3}$) in the receiving stream water (ANZECC, 2000). There was only a small reduction in the concentration of K by bark treatment. As yet there is no guideline for MPL for K concentration in surface water as it is not considered a significant water pollutant. Treatment with bark caused a considerable reduction in the concentration of TSS, BOD and COD in the effluent (Table 3). The BOD values in the bark treated effluent reached close to the recommended MPL value (5.0 g m$^{-3}$) in surface water. Bark treatment achieved almost complete removal of TSS, indicating that the bark material was an effective filtering medium for suspended solid and there was no indication of the breakdown of the bark material within the 6 months of period of the pilot-scale study.

**Advanced pond system**

Advanced pond systems (APS) have been in use for over 40 years, this technology however, has only recently been applied to the treatment of dairy farm wastewaters, particularly within New Zealand. APS comprise up to four different types of ponds, designed to optimise natural wastewater treatment processes (Craggs et al., 2004).

The first treatment step is anaerobic digestion, be it in a simple anaerobic pond, or a more advanced digester, where organic solids are microbially converted into methane and organic nutrients reduced into “plant-available” inorganic forms. The effluent from this stage is then discharged into “high rate ponds” that are shallow, and shaped to form a meandering channel raceway where the water is continuously mixed by a paddlewheel. Given their shallow depth (10-30cm), this high rate pond is virtually euphotic throughout its full depth, allowing algae to dominate. High rates of oxygen production occur as algae absorb dissolved nutrients during photosynthesis, and assimilate them into an algal biomass. The high rate of photosynthesis in these ponds causes elevated pH, which helps with nutrient removal by facilitating ammonia volatilisation and precipitation of phosphorus (Azov and Shelef, 1987; Nurdogan and Oswald, 1995). In addition to the actions of algae within this pond, heterotrophic bacteria utilise the available oxygen for aerobic breakdown of dissolved organic matter (Oswald, 1988).

An APS treating dairy shed effluent in Waikato Region of New Zealand, produced effluent with 50–60% less $BOD_5$, TSS, Total Kjeldahl N (TKN) and ammoniacal-N than equivalently sized two-pond systems (Craggs et al., 2004), with medians of 43, 87, 61 and 39 g $m^{-3}$ respectively. Total P was reduced by 70% to 19 g $m^{-3}$. Despite optimised conditions for P (especially, DRP) removal, i.e. high pH and enhanced algal assimilation, the amount of DRP in the final effluent remained high (15 g $m^{-3}$). Given the reliance of cations, in particular Ca, on the precipitation and thereby removal of DRP within the ASP system, greater DRP removal is likely to be achieved through the addition of Ca amendments and manipulation of existing Ca-P ratios in the high rate pond treatment. The faecal indicator bacterium, *Escherichia coli,* was reduced in the APS by two orders of magnitude to 918 most probable number (MPN) 100 $ml^{-1}$. In terms of overall performance, the optimised design of an APS gives considerably improved performance in the treatment of wastewater over two-pond systems.

## LAND APPLICATION OF DAIRY EFFLUENT

### Suitability of land application

The most common method of managing dairy shed wastes has been to return them to land as raw effluent directly from a collection sump or following some treatment in an effluent pond. Regional Councils encourage application of effluents to land as it is perceived to be less harmful to water quality than discharges directly to waterways, regardless of prior treatment processes. This encouragement has taken the form of short consent terms for discharges to surface water, but long consent terms for land discharges, resulting in very few lower North Island dairy farms (13 in Horizons, 0 in Hawkes Bay, 0 in Wellington, out of well over 1,000 farms in all three regions combined) now discharging effluent directly to surface water.

Land application of farm effluents has limitations: (i) it may not be possible when the soil moisture and climatic conditions are not favourable; (ii) requires greater pond storage facilities for holding the effluent when the soil moisture and climatic conditions are unfavourable for land application; (iii) can be difficult to manage and counterproductive when land is waterlogged; (iv) can contaminate groundwater and surface water by leaching and runoff of nutrients and pathogens; and, (v) the aerosols formed when spreading the effluent (e.g., piggery effluent) can result in odour problems (Monaghan and Smith, 2004; Wang et al., 2004; Houlbrooke et al., 2008; Luo et al., 2008; MAF, 2007).

Land area requirements stipulated by regulatory authorities for irrigating dairy shed effluent to pasture are most frequently specified in terms of N loading limits of 150 – 200 kg N $ha^{-1}$ (DEC, 2006). This is imposed primarily to minimise nitrate leaching, which is considered to be the major pollutant of groundwater systems. This approach has resulted in the impacts of other constituents of the effluent on the plant/soil/water system being overlooked. Furthermore while land application has been practiced for many years, the land application of dairy shed effluent in New Zealand is still a relatively new concept. As a result there is some difficulty in implementing sustainable land application systems as experienced amongst farmers is limited (Houlbrooke et al., 2004a).

### Effects on pasture

When managed correctly, soil helps to 'treat' or refurbish effluent in an 'environmentally-friendly' and sustainable manner while also providing a valuable source of nutrients to pasture.

This 'symbiotic' relationship will be successful only when the 'end-user', the dairy cattle, makes efficient use of the pasture, and when the rate of leaching of nutrients into groundwater is no greater than the leaching occurring beneath adjacent pastures that do not receive effluent application.

It can be argued that the policy of effluent application to pastures is being enforced without due regard to its impact on the quality of pasture as a feedstuff, for the sake of environmental gains. For example, the supply of large quantities of selective nutrients, such as N and K, through effluent irrigation could affect the nutrient balances and the botanical composition of the pasture (Campbell et al., 1980) and subsequently the herbage quality as a feedstuff. There have been claims that excessive nutrient loading through effluent irrigation has led to poor utilisation of pasture and nutrient-related metabolic disorders in grazing animals. Failure of the utilisation component decreases the efficiency of the land application system, and potentially increases in offsite environmental effects.

A field study was undertaken to examine the influence of dairy farm effluent irrigation on the botanical composition and nutrient concentration of pasture, and their combined effects on herbage quality (Bolan et al., 2004c). The pasture DM yield increased with increasing rate of effluent application, that could be attributed to the addition of both water and nutrients through effluent irrigation (Cameron et al., 1997). The yield response, expressed as kg DM $kg^{-1}$ N was less for effluent irrigation (8.5 kg DM $kg^{-1}$ N) than that for fertiliser N (12.6 kg DM $kg^{-1}$ N). Although the yield response values obtained in this field experiment were less than the maximum of 16 kg DM $kg^{-1}$ N obtained by others for intensively managed dairy pastures, the range was comparable (Crush et al., 1982; Roberts and Thompson, 1989; Ledgard et al., 1996). There is the potential that the difference in yield between fertiliser and effluent nitrogen additions could be a result of the form of nitrogen, with effluent nitrogen having a higher portion of organic nitrogen which is released over a longer time period (Longhurst et al., 2000).

The clover component in the pasture decreased slightly with increasing level of effluent irrigation. However, decreases were most pronounced with inorganic fertiliser N (29.6%) relative to effluent N (16.1%). It has often been observed that N application results in a decrease in clover content in legume-based pastures with a consequential decrease in biological N fixation (Crush et al., 1982; Ledgard et al., 1996). These studies have indicated that with increasing N addition, biological N fixation in legume-based pastures continued to decrease due to lower clover content, with decreases varying between 30 and 70% depending on the time of application, N level and the grazing management.

**Nutrient effects on animals**

In the study conducted by Bolan et al. (2004c), the N and K concentrations in pasture increased from 2.1% to 3.01% and from 2.2% to 3.6%, respectively, with increasing level of effluent irrigation. However, in the absence of Ca and Mg fertiliser application, the concentration of Ca and Mg in pasture decreased from 0.35% to 0.21% and from 0.26% to 0.13%, respectively with increasing level of effluent irrigation. Decreased uptake in the absence of Ca and Mg fertilisers can be attributed to excessive K loading under effluent irrigation (Lowe et al, 1994). High concentration of K in soil solution is likely to result in luxury uptake of this element by plants, particularly during winter and spring (Marschner, 1995). This results in the decreased uptake of other cations in order to maintain cation-anion charge balance in plants that generally leads to nutrient imbalances (McNaught, 1959; Lowe et al, 1994). It is also possible that uptake of N in the form of $NH_4^+$ from effluent irrigation may have caused deficiency of other cations (Kafkafi, 1990; Marschner, 1995).

The dietary cation-anion difference (DCAD) and grass staggers index (GSI) values, used as indices of the nutritive quality of pasture, were calculated (Wilson, 1999):

$$\text{DCAD} = ([Na^+] + [K^+]) - ([Cl^-] + [SO_4^{2-}]) \quad (1)$$

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

where: [ ] = milliequivalents $kg^{-1}$ DM

The DCAD value ranged from 220 – 740 meq $kg^{-1}$ DM, and increased with increasing level of effluent irrigation. It has been observed that DCAD values for most pastures in New Zealand range from 200 to 800 meq $kg^{-1}$ DM, indicating cation surplus over anions (Wilson, 1996). This normally results from the luxury uptake of K by pasture, resulting in high concentration of K in the herbage. Proper dietary cation-anion balance is important because animals attempt to maintain systemic acid-base balance and osmotic pressure in order to protect the integrity of cells and membranes and to optimise biochemical and physiological processes (Wilson, 1999).

When herbage with excess cations over anions (i.e., positive DCAD) is fed to animals, the concentration of alkali ions, such as bicarbonate ($HCO_3^-$), increases in body fluids resulting in alkalosis (Wilson, 1999). Conversely, when feedstuffs with a surplus of anions (i.e., negative DCAD) are ingested then the concentration of acidic hydrogen ions increases and metabolic acidosis occurs. There have been conflicting reports on the optimum level of DCAD value required for dairy cattle (Roche, 1997). It has, however, been shown that feedstuffs with high DCAD values tend to increase the incidence of milk fever, particularly in springing cows and recently calved cows, and that supplementation of pre-calving rations with anionic salts with low DCAD values reduces the incidence of milk fever (Beede, 1992; Wilson, 1996).

The GSI value ranged from 1.35 to 3.46 and increased with increasing level of effluent irrigation. Application of gypsum and Epsom salt resulted in a decrease in DCAD and GSI values mainly due to an increase in the concentration of Ca and Mg in pasture. When GSI values of the pasture exceed 2.2, the risk of hypomagnesaemia (grass staggers) development is enhanced (Mason and Young, 1999). This condition is generally linked with animal serum Mg levels less than 10 – 15 mg $litre^{-1}$, compared to normal level of 17 – 30 mg $litre^{-1}$ in cattle. This arises in response to diets inherently low in Mg content that can then be confounded by K-induced inhibition of Mg absorption in the rumen (Grunes and Rending, 1979; Roberts, 1994). The deficiency of Ca and Mg is exacerbated by high concentrations of K in the herbage, occupying most of the metallothionein protein in the rumen and leading to K-induced excretion of other bases.

The risk of metabolic disorders developing in cattle may be mitigated by avoiding grazing of effluent disposal areas, particularly by springing or calving cows. Where grazing does occur, K levels in soil and pasture can be monitored by routine soil and herbage testing, and Ca and Mg based fertiliser applied as required along with a reduction/exclusion of K based fertilisers. Where pasture high in K is ingested, an accompanying supply of magnesium-based supplements to cattle may be required (Bolan et al., 2004c; Lowe et al, 1994).

## Land area requirement

Based on typical nutrient composition values of effluent, an application area of approximately 4 ha per 100 cows is generally sufficient to meet Regional Council N loading requirements (200 kg N $ha^{-1}yr^{-1}$), however in light of concern regarding excess K loading, best management practice recommends extended disposal areas of approximately 8 ha per 100 cows thereby reducing K loading and the risk of metabolic disorders in grazing animals (DEC, 2006). This approach, however, has the effect of reducing the effluent nitrogen input to 100 kg N $ha^{-1}$ $yr^{-1}$, which is significantly less than what may otherwise be applied to non-effluent pastures as synthetic fertiliser. Some Regional Councils disallow the application of urea to effluent blocks, so the farmer's choice is either to run his effluent pastures in nitrogen deficit, or to risk metabolic disorders in his herd.

## Method of application

Travelling irrigator systems are the most widely used effluent application system on New Zealand farms. Low application rate sprinklers, which are smaller in size (e.g. K-line Pods), are becoming more prevalent and have certain advantages. For instance they are easily deployed by hand, are capable of applying at lower rates than traditional travelling irrigators and generally require limited financial outlay (Houlbrooke et al., 2008). There have been heroic failures with attempts a decade ago to irrigate effluent onto North Otago pastures using border dyke infrastructure, and at the other end of the scale there are now some very sophisticated solid set impact sprinkler systems, capable of applying low application rates. However, there are situations where irrigation from a tractor-drawn tanker provides the best balance between convenience, cost, and meeting regulatory requirements within the limits of an unforgiving landform setting. The use of tankers is often preferable when the material has a think consistency, which can be difficult to pump and spread using conventional irrigation technology.

## Irrigation integration with ponds

Travelling irrigation systems typically pump effluent from a shed sump straight onto the pasture. The treatment ponds that were widely used until recently have mostly been filled in as a result to Regional Council to abandon surface water discharge in favour of land application. However, soil moisture conditions need to be considered prior to irrigation in order to avoid excess leaching of nutrients, pathogens and effluent constituents under saturated soil conditions (Houlbrooke et al., 2008). Well maintained and appropriately sized effluent storage facilities can enable irrigation to be deferred during periods of high soil moisture content so as to avoid poorly timed irrigation events (Houlbrooke et al., 2004b) and are now variously promoted or required by Regional Councils as conditions on resource consents. These ponds are typically for storage rather for any treatment, and some Regional Councils are requiring that they have very low leakage rates ($<10^{-9}$ m $s^{-1}$). While some clayey soils meet this permeability requirement when compacted, most require durable plastic liners to be used.

The concept of deferred irrigation is a valuable tool to assist with maximising the nutrient value of the effluent and assisting with minimising the environmental effects associated with leaching. It should be noted that deferred irrigation differs from deficit irrigation, in that deficit irrigation does not allow for drainage following irrigation while deferred irrigation tries to minimise the effects of drainage following irrigation.

Where storage ponds have not yet been installed, the requirement to irrigate as effluent is accumulated runs risks of ponding or runoff. This on its own can provide an incentive for some large herd operators to minimise their shed water use. This in turn increases both effluent consistency and nitrogen loading, which in turn challenges both the durability of irrigation reticulation and the ability of the equipment to apply effluent at a low enough rate to meet consented nitrogen limits. When ponds are installed to buffer against periods of no land application, consideration has to be given to managing the solids in the effluent, which can precipitate out and accumulate in the base of the ponds. This in turn reduces the pond's retention ability if the sludge levels area not monitored.

**Solid separation**

Where long reticulation distances and/or heavy effluent consistency present a challenge, solids separation either by screw press or by weeping wall have been useful. The separation of solids can bring a disappointingly small proportion of the total nitrogen load with them, allowing very little reduction in the area of land required to satisfactorily receive the liquid effluent loading rate. However, the separation of solids, especially from effluent of a heavy consistency, pays dividends in the mechanical ease of irrigation. Low tech weeping walls have much to offer in this regard, with no moving parts to break down, but the design needs to be right, they are hard to adjust once the concrete has been poured. Weeping walls are popular in Southland, but still uncommon in the lower North Island.

**Nutrient modelling**

Under the Dairying and Clean Streams Accord (2003), a joint agreement between Regional Councils, Central Government and New Zealand's main dairy processor, Fonterra, dairy farmers providing milk to Fonterra are required to use nutrient budgeting to manage on and off farm nutrients. Furthermore, many Regional Councils in New Zealand now require nutrient budgets as part of the resource consent application procedure for discharging dairy shed effluent. In developing nutrient budgets for farms, a number of site specific influences, such as soil type, irrigation depth, management techniques, local climate must be considered (Snow et al., 1999). Computer simulation models that encompass a wide range of variable parameters are used to help predict and understand the flow of water and nutrients in soil-plant systems and are increasingly being used for decision support when developing on-farm nutrient management plans, and help demonstrate environmentally sound farming practice to local communities and international customers of agricultural products.

A number of models, for instance OVERSEER® (AgResearch Ltd), LECHM (Wagenet and Hutson, 1989) and APSIM (Agricultural Production Systems Research Unit, 2007) also have the capacity to simulate the fate of nutrients and water during effluent irrigation. OVERSEER® nutrient budgeting model is one of the main nutrient management tools used in the New Zealand farming industry (Wheeler et al., 2008). It enables users to develop budgets for N, P, K, S, Ca, Mg, Na and H, as well as greenhouse gas emissions on a block (paddock) or whole farm basis. Model predictions are usually supported and validated by routine soil sampling and can be used to minimise nutrient loss and maintain crop quality.

Perceptions by river users of declining water quality have become significant social drivers for improving the environmental results of the management of farm dairy effluent. Starting with the Dairying and Clean Streams Accord (2003), and continuing with better focused resource consent conditions and most recently with more aggressive enforcement action by some Regional Councils, social preferences and pressures have raised the bar for the off-farm environmental effects of effluent management. And as with all imposed changes, it is important not to lose sight of the costs involved, or on whom they fall.

## CONCLUSIONS

In New Zealand, farm effluents that were traditionally treated biologically using two-pond systems and then discharged to streams are now usually irrigated onto pastures, with or without the use of a storage facility. Effluents discharged to streams now require additional treatment to minimise the concentration of nutrients that may otherwise lead to eutrophication of water-ways. The advanced pond treatment system and use of materials capable of absorbing effluent constituents are two effective methods whereby nutrient concentrations can be reduced, where discharge to water is to be pursued.

Greater appreciation of the nutrient value of farm effluents has also lead toward land application that facilitates the re-cycling of valuable nutrients and water, and is regulated by regional councils to avoid groundwater and surface water pollution. Where effluents are returned to land there is a need to manage nutrients within the overall nutrient budgeting of the farm operations to ensure forage quality and protect against off-site environmental pollution.

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