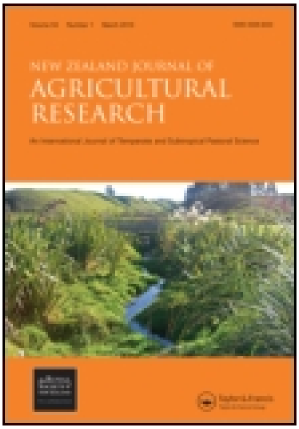

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Effects of land application of farm dairy effluent on soil properties: A literature review

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Effects of land application of farm dairy effluent on soil properties: a literature review

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Abstract Land application of wastes is becoming an increasingly popular practice. The application of farm dairy effluent (FDE) to land, as opposed to direct discharge to waterways, is the preferred mechanism for disposal in New Zealand as regulatory authorities move to protect and enhance water quality and meet Maori spiritual and cultural values. For example, in the Waikato the percentage of dairy farmers who apply FDE to land has risen from 35% in 1993 to effectively 100% in 2004. Land application recognises the nutrient value of FDE; however, it is not without risks. At present the management of land application of FDE is primarily concerned with potential nitrate contamination of groundwater. There are other, considerable, potential risks such as microbial contamination and impacts on soil properties. Despite soil properties being spatially extremely variable and the effects of land application of FDE being long-term, the majority of studies on the impacts of FDE to soil properties are limited in their temporal and spatial extent. The evidence of effects of FDE on soil properties is limited; however, it is reviewed and linked to the effects of land application of other effluents in general and their effect on specific soil properties. The effects of FDE application on soil physical properties are highly variable (e.g., both increases and decreases in soil hydraulic conductivity have been observed). This is in part because of measurement issues and the difficulty in separating out the effects of the FDE application from the effects of stock management. Overall, most changes due to FDE application appear to improve the soil's long-term fertility, i.e., increased concentrations of total nitrogen, total phosphorus, and plant available nutrients. FDE application has been observed to result in a greater and more diverse microbial biomass. However, optimal environmental management and the continued use of the soil for FDE management require ongoing monitoring.

Keywords farm dairy effluent; soil properties; pasture; land application; nitrogen; phosphorus

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INTRODUCTION

Dairy farming is crucial to the New Zealand economy: dairy products constituted 17% of the total export earnings in 2003 while agriculture contributed 4.9% of GDP (Statistics New Zealand 2004a,b). Over the past 10–15 years dairy farming in New Zealand intensified dramatically: total cow numbers have increased from 2.4 million to 3.7 million (55% increase) between 1990/91 and 2002/03 (Livestock Improvement Corporation 2003), the average stocking rate has increased from 2.1 to 2.6 cows per hectare in the 20 years to 1997/98 (Ministry for the Environment 1999), and, associated with this increase in stock numbers has been an increase in the use of nitrogen (N) fertiliser (Parliamentary Commissioner for the Environment 2004). However, the increasing number of livestock, and the associated processing industries, pose a considerable threat to the environment due to the need to safely dispose of the waste generated.

Farm dairy effluent (FDE) is the mixture of dairy cow faeces and urine deposited during milking and subsequently diluted with wash-down water during the cleaning of the milking area and the associated holding yards. Thus FDE is a very dilute organic effluent composed of a soluble fraction and an organic solids fraction; the solids content is generally <1% (Longhurst et al. 2000; Barkle et al. 2001).

Irrigation of FDE onto pasture is increasingly being recognised as a means for biological treatment and recognises the fact that FDE is a resource to be utilised for its mineral content rather than a waste for disposal. Land application of FDE can provide valuable minerals and organic matter for pasture. However, inappropriate application rates or timing can lead to poor utilisation by plants, causing nitrate leaching and groundwater contamination; surface water contamination; waterlogging of soils; soil nutrient imbalances and/or animal health problems (Wang et al. 2004). In New Zealand, land application of FDE has become the preferred treatment method to minimise the risk of contamination of surface waters and to address cultural concerns about the addition of waste material to waterways. For example, since 1993 the proportion of Waikato farmers who irrigate FDE onto pasture rose from 35% to nearly 70% in 1997 to effectively 100% in 2004 (Barkle et al. 2000).

The challenges of effluent application are well recognised. The purpose of this paper is not to present a comprehensive review of effluent irrigation, which may be found elsewhere (McLaren & Smith 1996; Cameron et al. 1997; Bond 1998). Effluent is a broad term that includes sewerage sludge and wastewater from industrial plants, such as dairy factories and slaughterhouses, and FDE. Recent publications have addressed aspects of effluent application, such as: (a) the effects of industrial and sewerage effluents on soil biochemical properties (Speir 2002); (b) the effects of dairy factory effluent on particular soil properties (Degens et al. 2000; Cameron et al. 2002, 2003); and (c) the relationship between FDE application and groundwater quality (particularly related to nitrate leaching (Di et al. 1998a,b; Close et al. 2001; Di & Cameron 2002a,b). The purpose of this paper is to address the effects of FDE application on soil properties in the context of the Resource Management Act (RMA), which places an emphasis on sustainable environmental management and, among other things, the safeguarding of the life-supporting capacity of waterbodies and the soil. The context for these effects is that relative to the 1993 situation, using average concentrations, there is now an extra 16 million cubic metres (approximately) of FDE in New Zealand per year, containing 4300 tonnes of N and 1100 tonnes of phosphorus (P) (Houlbrooke et al. 2004).

FARM DAIRY EFFLUENT MANAGEMENT

Prior to the growth in land application of FDE, the effluent from New Zealand dairy sheds was typically treated in a two-stage pond (anaerobic followed by aerobic) system prior to discharge to a surface waterway (Zaman et al. 2002). For example, during 1996/97 about 35% of farms in the Waikato region were using this waste treatment system (Longhurst et al. 2000). Various studies (e.g., Cooke et al. 1979; Hickey et al. 1989) have been conducted on the quality of final discharge because of its impact on the environmental pollution of streams. From a mineral recapture perspective, and as a means to reduce environmental effects on surface waterways, FDE application to land is of greater value than utilising effluent from oxidation ponds (Longhurst et al. 2000). While effluent irrigation enables diversion of nutrients from waterways, it potentially results in other environmental problems if not carefully managed. These include increased recharge to the groundwater accompanied by salts and nitrate, accumulation of salt and other minerals (such as P) in the soil, and the increased risk of runoff of these contaminants into waterways.

Scientists have recognised the potential of New Zealand soils to be used as sites for effluent

Table 1 FDE composition.

	Total N (mg litre^{-1})	Ammonium-N (mg litre^{-1})	Nitrate-N (mg litre^{-1})	Organic N (mg litre^{-1})	Total P (mg litre^{-1})
Cooke et al. (1979)[1]	167/198	13.5/26	2.2/2.3		16/19.5
Goold (1980)[2]	195 (56)				58 (17)
Di et al. (1998b)[3]	363 (199)	95 (49)	0.5 (0.6)		40 (17)
Silva et al. (1999)[4]	240	61	0.19		
Longhurst et al. (2000)[5]	269 (181–506)	48 (13–132)	2 (1–6)	219 (144–374)	69 (21–82)
Singleton et al. 2001[5]	90 (44–186)	23 (5–70)	0.05 (<0.05 – 0.45)		67 (34–125)

[1]Run1/run3.
[2]Mean values (standard deviation).
[3]Mean (standard deviation) of five samples.
[4]Mean of four samples.
[5]Mean values (range).

application (e.g., Wells 1973; Childs et al. 1977); however, this assessment has historically been based upon assuming that the soil acts as a "filter", that crop health is maintained, and that groundwater and surface water quality are not adversely affected. Gradual adverse changes in soil properties, such as hydraulic conductivity, leaching of nutrients, plant productivity and structural integrity have been seen as less important. At present, obtaining a resource consent for FDE application to land is focused on ensuring: the chosen site has appropriate soils and land area to adequately cope with the hydraulic load; the application design minimises soil saturation and potential bypass flow; and the loss of nutrients (especially N as nitrate) is minimised. That is, there is little regard for the broader impacts of FDE application as it is largely considered as a simple addition of N (and in many cases no different from N application as fertiliser despite its quite different properties).

The recent large increase in dairying in Southland has been anecdotally blamed for the observed increased levels of microbial contamination (McLeod et al. 2003). Hence, FDE management needs to consider soil properties because the transport of pathogenic microoganisms found in FDE, such as bacteria, protozoans and viruses, through soil is affected by soil properties (Dean & Foran 1992; Aislabie et al. 2001). In particular, FDE application may affect soil hydraulic conductivity and the potential for macropore/bypass flow and so the transport of pathogenic microorganisms to groundwater or surface water. Despite this there are few long-term datasets, or data from a wide range of locations, on the composition of FDE.

FARM DAIRY EFFLUENT

Compositional analysis of FDE has been undertaken by a number of researchers (e.g., Cooke et al. 1979; Goold 1980; Longhurst et al. 2000; Table 1). Compositional variations are likely due to the time of milking, the age and breed of the herd, farm fertiliser policy, feed quality, washwater management and the time relative to lactation. The primary concern has been the concentrations and form of the N component of this effluent, due to the potential for nitrate leaching and groundwater contamination. However, from water quality and soil properties points of view the composition of FDE in terms of all its constituents is important.

Typically 60–85% of the total N present in the effluent is in organic form (Selvarajah 1996; Barkle et al. 2001). Most of this organic N is from the faecal material. The soluble fraction includes any soluble material from the faecal material plus urine. This urine contributes some organic compounds; 60–90% of the total N in urine is urea, which is rapidly hydrolysed to ammonium-N. The exact composition of FDE is spatially and temporally variable. For example, Longhurst et al. (2000) found that, across 284 analyses from a range of locations, between 1977 and 1997, the mean total N concentration in FDE doubled from c. 200 to 400 mg litre^{-1}; this increase is likely to have continued as dairy herd size increases because the volume of washwater used per cow decreases. Within this general trend is considerable spatial variability; that is, across the 11 studies used by Longhurst et al. (2000) the mean total N content varied between 181 and 506 mg litre^{-1} while the standard deviation varied between 47 and 461 mg litre^{-1}. Goold (1980) examined chemical composition of dairy shed effluent from a 53 ha dairy unit milking between 102 and 130 cows from 1972 to 1976. In addition to reporting N and P concentrations (Table 1) the concentrations (mg litre^{-1}) of major elements were reported, mean (standard deviations): magnesium (Mg) 39(11); calcium (Ca) 177(50); sodium (Na) 54(14); and potassium (K) 435(160), i.e., considerable temporal variability. Unlike N, Toor et al. (2004) reported that the majority of the P in FDE was inorganic (86%) while organic P was only 9.6%.

Hence, FDE is composed of a mixture of N and P (in a range of forms), organic carbon (C), sulphur (S), Ca, Na, K and Mg; a solution that has a range of possible pHs and salinities and that may contain enteric pathogens. Therefore, there are a wide range of possible effects of land application of effluent. Added to the complexity of the changes in FDE composition is the lack of understanding of the interaction of these inorganic and organic, soluble and solid, fractions during the decomposition of FDE; particularly when combined with soil which is also highly variable in its physical, chemical and biological properties.

As noted by Spier (2002) in his review of changes in soil biochemical properties under effluent irrigation, a major problem is the time period (c. 20 years) between the earliest and latest published reports. Over this period methods have changed, properties analysed changed, and totally new concepts and methods have appeared. For example, Cooke et al. (1979) analysed nitrate-N and ammonium-N by distillation whereas Di et al. (1998b) used flow injection analysis or ion exchange chromatography.

Only recently has attention been focused on the potential for microbial transport through soil (Aislabie et al. 2001). While some areas have been subject to intensive investigation (e.g., Canterbury and Waikato), New Zealand-wide information is scarce. Thus, it is very difficult to compare results and draw conclusions. While effluent from a range of sources may be applied, and changes in soil properties noted, the composition of effluent is variable and the changes observed may be site specific. However, despite difficulties with generalising results and conclusions, it is important to try.

NITROGEN

The application of FDE is usually associated with additions of N to the soil surface. In many countries groundwater and surface water have become severely contaminated by nitrate-N as a result of both excessive application of N fertilisers and land application of nutrient-rich wastes (Carter 1997). Nitrate-N concentrations in waterbodies are a problem because of their potential impact on human health and ecology (Close et al. 2001). Hence, the New Zealand Ministry of Health has set a maximum limit for nitrate-N in drinking water (Ministry of Health 2003). The Australia New Zealand Environment and Conservation Council has set guidelines for stock water requirements and recreational water (ANZECC 2000). The loading rates permitted by many regional councils for FDE are commonly written in terms of the addition of N.

As nitrate-N is highly soluble, the setting of guidelines for the application of FDE has attempted to recognise that the amount of N input should be related to the expected plant uptake or capacity of the soil to absorb N. Hence, there are numerous studies examining the controls on nitrate-N leaching and the appropriate guideline values of both fertilisers and FDE in New Zealand (Ledgard et al. 1996a; Di et al. 1998a,b; Silva et al. 1999) and overseas (Jarvis et al. 1987; Unwin et al. 1991). Results from Di et al. (1998a,b) suggest that the amount and timing of potential nitrate leaching is not a simple function of the N input because the form of the N affects the potential for leaching. Two examples demonstrate this at different scales. In the study by Cameron & Di (2004) the N in pig slurry was in a form (ammonium-N) that was leached much more rapidly compared to the less available organic N form present in the dairy pond effluent. At the small scale, urine is recognised as potentially having a major impact because it is a concentrated solution of N in a readily leached form deposited in a relatively small area (Ball et al. 1979; Sherwood 1986; Silva et al. 1999).

Nitrogen from grazing animals (sheep, deer or cattle) is in the form of urea (H_2NCONH_2) from urine or organic N in faecal materials. These are converted to ammonium-N through hydrolysis reactions (urea; i.e., $(NH_2)_2CO + 2H_2O \rightarrow (NH_2)_2CO_3 \leftrightarrow NH_4^+ + NH_3 + CO_2 + OH^-$) or decomposition (faecal material by mineralisation, a series of reactions leading to the breakdown of complex proteins into amino acids. Soil organisms cause these amino compounds to be converted into ammonia, i.e., $R\text{-}NH_2 + H_2O \rightarrow NH_3 + R\text{-}OH + \text{energy}$). Nitrogen transformations at the land surface, in the soil zone, and vadose zone are complex (McLaren & Cameron 1996; Close et al. 2001). Effluent contains little or no nitrate and, therefore, it contributes only indirectly to soil nitrate concentrations following a series of microbial transformations occurring in the vadose zone. Mineralisation of organic N to ammonium-N (mineralisation) and a two-stage nitrification process that oxidises ammonium-N to nitrate-N (i.e., $2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + \text{energy}$ and $2NO_2^- + O_2 \rightarrow 2NO_3^- + \text{energy}$) are necessary before effluent N contributes to soil nitrate-N (Williamson et al. 1998). While conceptually understood, the relative importance of each of the interactions is not easy to quantify because of site-specific controls such as the amount of N applied, the climate, the type of pasture, the proportion of the N applied that is organic N (which only decomposes slowly) and the impact of stocking through its effect on urine additions. Nonetheless, Di & Cameron (2002a,b) propose a method of estimating N leaching losses and critical N application rates for pastoral soils using minimal input data. They have suggested that volatilisation and denitrification are of only minor importance; however, soil texture and antecedent conditions can affect these processes (Di et al. 1998a; Barton et al. 1999). Soil texture has also been noted to affect the rate of N mineralisation (Strong et al. 1999; Stenger et al. 2001). To further complicate matters Singleton et al. (2001) noted that in addition to nitrate-N a considerable proportion of the total N leached could be dissolved organic N.

In addition to the potential for nitrate-N leaching, effluent irrigation can decrease total soil N (Falkiner & Smith 1997; Sparling et al. 2001), or result in no change (Schipper et al. 1996; Williamson et al. 1998; Sparling et al. 2001). However, most researchers have shown that effluent irrigation increases total soil N (Quin & Woods 1978; Ross et al. 1982; Speir et al. 1987; Ross et al. 1989;

Carey et al. 1997; Magesan et al. 1999; Barkle et al. 2000; Degens et al. 2000; Peacock et al. 2001; Cameron et al. 2002; Hawke & Summers 2003). For example, Barkle et al. (2000) in their lysimeter study, using Waikato Te Kowhai silt loam, observed that 3 years of FDE irrigation at a loading rate of 3900 kg N ha^{-1} resulted in a significant increase in total N. Typically 60–85% of the total N in FDE is in organic form, which is not immediately available for plant uptake. This organic fraction can be conceptually subdivided into an easily mineralisable pool that becomes available within a year of application, and a more resistant pool with a mineralisation rate closer to that of native soil organic matter (SOM). Thus, while the accumulation of total N is usually viewed as positive, the rate at which this resistant organic matter from FDE accumulates and the effect of any accumulation on other SOM-related pools, such as microbial biomass, needs to be determined to ensure the quantification of the effects of the long-term impacts of FDE application and sustainability of applying FDE to land.

Effluent irrigation has also been shown to affect pasture botanical and chemical composition (Ledgard et al. 1982; Saunders 1984). In particular, the addition of N via urine resulted in a decline in clover and an increase in grass and clover N, and cations such as Ca (see below).

CARBON

Effluent irrigation can decrease total soil C (Falkiner & Smith 1997; Sparling et al. 2001); or cause no change (Ross et al. 1989; Schipper et al. 1996; Liu et al. 1998; Degens et al. 2000; Sparling et al. 2001). However, most researchers have shown that effluent irrigation can increase total soil C (Quin & Woods 1978; Ross et al. 1982; Stadelmann & Furrer 1985; Spier et al. 1987; Holford et al. 1997; Barkle et al. 2000; Peacock et al. 2001; Redding 2001; Zaman et al. 2002). For example, Barkle et al. (2000) noted that 3 years of FDE application at a loading rate of 42 000 kg C ha^{-1} resulted in a significant increase in organic carbon to a depth of 20 cm (the maximum depth monitored). Increases in the SOM has been correlated with increases in cation exchange capacity (CEC). The soil CEC depends on the amount of humus in the soil, the amounts and types of clay mineral present, and the pH at which the determination is made (McLaren & Cameron 1996). Parfitt et al. (1995) measured the CEC for 347 A-horizons and 696 B-horizons of New Zealand soils at pH 7; the results showed that most of the CEC arises from soil organic mater (significant at $P < 0.001$).

The additions of N and C are also important if they change the C:N ratio of the soil. The addition of C, which does not occur under N fertiliser application, stimulates the soil microbial pool and induces enhanced N immobilisation, and possibly denitrification (Ghani et al. 2005). The critical value above which N immobilisation occurs is usually 20:1 (McLaren & Cameron 1996). A decrease in the C:N ratio makes greater quantities of the mineralisable forms of soil N available for plant uptake.

PHOSPHORUS

Irrigation by piggery effluent, such as in Australia, has shown that the long-term application of piggery wastewater may encourage leaching of N and P in soils with limited capacity to retain these nutrients (e.g., Weaver & Ritchie 1994; Redding 2001; Philips 2002a,b). In some cases soils containing elevated levels of P can act as local point sources for pollution of groundwater and surface water. Piggery effluent also tends to be highly alkaline and has concentration of ammonium-N, which is rapidly oxidised to nitrate-N, and so nitrate-N leaching is a major issue.

In general, New Zealand soils have a high capacity to absorb P and hence effluent irrigation has been observed to increase soil total P (Quin & Woods 1978; Spier et al. 1987; Liu et al. 1998; Nair et al. 1998; Menzies et al. 1999; Spier et al. 1999; Degens et al. 2000; Redding 2001; Hawke & Summers 2003). Field monitoring (e.g., Falkiner & Polglase 1997; Menzies et al. 1999) has shown that P adsorption curves developed in the laboratory are poor indicators for P adsorption in the field, and hence unsuitable for the prediction of P sorption capacity and the life of land treatment facilities. Siddique & Robinson (2003) have shown that the magnitude of changes in P sorption and availability, due to application of P from a range of organic sources such as poultry manure, sewerage sludge, and cattle slurry were affected by the source of the P.

While there are studies on the effects of effluent irrigation of soil P levels (e.g., Quin & Forsyth 1978; Quin & Woods 1978; Spier et al. 1987; Tomer et al. 1997; Spier et al. 1999), there are few on the effects of FDE. Dairy cows are recognised contributors to nutrient cycling in grazed pastures by returning large amounts of P, K, Ca, Mg, Na, and S in dung (Aarons et al. 2004a,b). Aarons et al. (2004a,b) suggest that the mechanism for the movement of P from dung into the soil was via physical incorporation, and that this dung P contributed to plant-available soil inorganic P. Furthermore, Saunders (1984) noted that the higher P levels in the soil associated with dung and urine application resulted in changes to pasture composition and pasture chemical composition.

In one of the few New Zealand studies on the effects of FDE application on soil P, Toor et al. (2004) outlined the results from lysimeter and field experiments to draw conclusions about impacts of P inputs on soil P amounts and forms. While the applied FDE was high in inorganic P (86% versus 9.6 organic P), the leachate collected contained mainly organic P (85–88%). The results suggested that the inorganic P applied in FDE was adsorbed because of the high P fixation capacity of Lismore subsoil. Therefore, in addition to the potential for FDE application to result in organic P leaching losses and increased soil P, it appears likely that, over time, the number of soil P fixation sites may be saturated due to high P inputs.

OTHER ELEMENTS

While effluent from meatworks (Spier et al. 1987) and piggery effluent (Phillips 2002b) have been observed to lead to a decline in soil cation exchange capacity and/or exchangeable cations (K, Na, Ca, Mg), most studies have shown that effluent irrigation can increase soil cation exchange capacity and/or exchangeable cations (K, Na, Ca, Mg) (Quin & Woods 1978; Goold 1980; Stadelmann & Furrer 1985; Bernal et al. 1992; Tomer et al. 1997; Liu et al. 1998; Menneer et al. 2001; Menzies et al. 1999; Phillips 2002b; Hawke & Summers 2003). Goold (1980) and Hawke & Summers (2003) both note an increase in CEC and exchangeable cations following FDE application. The overall results are complicated by studies, such as Philips (2002b), which showed that Ca and Mg decreased in the upper layers of the profile, due to leaching, at the same time K and Na concentrations increased. This has been attributed to the typically high concentrations of K in animal manure, inducing the leaching loss of other exchangeable bases due to the competition with K (Bolan et al. 2004). Increased K concentrations can detrimentally affect soil physical properties because high levels of K may disperse clays. Aarons et al. (2004a,b) attributed changes in soil nutrient concentrations due to dung decomposition to the solubility of the individual nutrients; hence, whereas most P was physically incorporated, K was leached into the soil. Changes in the balance between K, Ca, and Mg concentrations in the soil can also affect pasture composition and animal health.

pH

A critical control on soil chemical properties, and changes to these properties, is on soil pH. Given effluent is usually alkaline, and most New Zealand soils are acidic, most studies have shown that effluent irrigation increases soil pH (Quin & Woods 1978; Goold 1980; Saunders 1984; Spier et al. 1987; Schipper et al. 1996; Holford et al. 1997; Kim & Burger 1997; Menzies et al. 1999; Barkle et al. 2000; Redding 2001; Sparling et al. 2001); however, in a few cases effluent irrigation has decreased soil pH (Bernal et al. 1992; Liu et al. 1998; Zaman et al. 1998).

Soil electrical conductivity

Associated with the change in soil pH is the effect of effluent irrigation on soil electrical conductivity. This has been particularly important in Australia due to the dry conditions and potential increases in salinity (Falkiner & Polglase 1997; Bond 1998). In New Zealand, Menneer et al. (2001) reported increases in soil salinity following the application of Na-contaminated wastewater. Significantly, this change in soil chemical properties was linked to changes in soil physical properties. There was no evidence of dispersed clays in the leachate and so it appeared that organic matter dissolution was affecting the collapse of aggregates and the increase in soil electrical conductivity and the exchangeable Na percentage was leading to a decline in hydraulic conductivity. Whereas Cameron et al. (2003) in their study of the effect of dairy factory effluent irrigation, noted that while the Na in the dairy factory effluent decreased the wet aggregate stability and the hydraulic conductivity, the effect of the additions of organic carbon enhanced the wet aggregate stability and reduced the risk of structural degradation, and a decline in hydraulic conductivity. Hence, changes in soil chemical properties are intimately linked to changes in soil physical properties.

CHANGES IN SOIL PHYSICAL PROPERTIES

As in the Cameron et al. (2003) example, effluent irrigation has been linked to changes in hydraulic conductivity. Irrigation with slaughterhouse effluent has been observed to enhance clay aggregation (Churchman & Tate 1986), while the irrigation of a range of organic wastes, such as compost, sawdust, and poultry manure onto tropical soils, resulted in increases in total porosity, macroporosity, saturated hydraulic conductivity, and water retention (Mbagwu 1989). Similarly, the addition of sewerage sludge and compost to an Italian sandy loam increased total porosity and the stability of soil aggregation (Pagliai et al. 1981). Likewise, the irrigation of dairy factory effluent affected a range of physical properties of Horotiu and Te Kowhai (Waikato) soils: the hydraulic conductivity of both soils was increased; the bulk density of the Horotiu soils decreased, whereas the Te Kowhai soils exhibited no change; and the macroporosity of both soils was not affected (Sparling et al. 2001).

While the additions of organic matter via FDE irrigation can result in enhanced soil structure, effluent irrigation can result in plugging of pores (Clanton & Slack 1987); changes in the pore size distribution of the topsoil (Cook et al. 1994); a decrease in soil hydraulic conductivity, due to increased soil microbial biomass (Magesan et al. 1999); a decrease in soil hydraulic conductivity and a change in micropore and macropore flow, due to aggregate collapse (Menneer et al. 2001); or no change in soil physical properties, presumably due to only short-term effluent irrigation (Spier et al. 1999). Other soil physical properties can also be affected, particularly by long-term effluent application. For example, effluent irrigation can decrease soil bulk density (Bernal et al. 1992; Sparling et al. 2001); however, changes in soil physical properties are difficult to quantify because they tend to occur only over the long term and soil physical properties are notoriously variable and difficult to measure with a high degree of accuracy, repeatability and precision. The measurement issue is particularly problematic if the experimental approach is one of an "affected" site and a "control" site. In addition, there may be seasonal variation in soil physical properties and the effects of FDE irrigation have to be separated from the effects of stock management (Monagahan et al. 2005).

SOIL BIOLOGICAL PROPERTIES

As identified above, changes in soil chemical and physical properties are affected by and affect soil biological properties. The mineralisation of FDE results from a sequence of different microbial and extracellular enzyme activities; hence, the microbial turnover of FDE is affected by the FDE loading rate (Barkle et al. 2000, 2001). Not only is the total soil microbial biomass affected by FDE, but also the addition of readily available C can result in a more diverse and dynamic microbial system than inorganically fertilised soil (Peacock et al. 2001). Studies conducted in Canterbury have shown that slaughterhouse effluent markedly increased microbial biomass, enzyme activities and net soil mineral N production from SOM (Ross et al. 1982). While additions of FDE significantly increased the gross N mineralisation rate, and protease, deaminase, and urease activities, the addition of NH_4Cl did not increase the gross N mineralisation rate (Zaman et al. 1999a). However, these results are complicated by the associated field experiments (Zaman et al. 1999b) which showed that the addition of FDE to the soil resulted in significantly higher gross N mineralisation rates than the addition of NH_4Cl, but NH_4Cl still increased the gross N mineralisation unlike during the laboratory experiments (Zaman et al. 1999a). Also the soil water potential has an effect on gross rates of mineralisation and nitrification (Zaman et al. 1998). Ghani et al. (2005) further showed that regular applications of dairy factory effluent encouraged N immobilisation, particularly at low temperatures because the regular supply of soluble C prolonged microbial immobilisation; however, at higher temperatures enhanced soil microbial activity may respire most of the C within a short time and therefore minimise the negative effects on nutrient availability for plant growth.

OTHER EFFECTS

Farm effluent application generally increases pasture yield; however, the effect is a function of the application rate and method, season, soil fertility, and climatic conditions. While the application generally increases the dry matter yield, the composition of the botantical material may change; typically the clover component of a white clover/ryegrass pasture decreases with additional applications of N (Bolan et al. 2004; Wang et al. 2004; Monaghan et al. 2005).

Farm effluents typically contain high concentrations of K. In addition to excess K accumulating in the soil or being leached from the soil profile, luxury uptake by plants can result, which can considerably increase K uptake by animals. This can potentially lead to a nutrient imbalance in dairy animals and adverse effects (Bolan et al. 2004; Wang et al. 2004).

DISCUSSION

From the point of improved water quality, Houlbrooke et al. (2004) concluded that because 80–98% of the nutrients applied in the FDE were trapped by the soil land treatment, a considerable reduction in the quantity of nutrients reaching freshwater bodies, land treatment could have considerable positive effects. The nutrient content of FDE means it is a potential fertiliser (Roberts et al. 1992); however, the use of FDE as a fertiliser requires the nutrient requirements for biologic activity to be known. This knowledge allows recommendations to be made on the rates of FDE application together with any supplementary solid fertiliser that might be required. The dilute nature of FDE means that land application has the further benefit that it is a mechanism for applying water, which may further enhance pasture growth and limit the need for (often) expensive irrigation water. However, successful utilisation of FDE requires application rates to be balanced with the nutrient needs of the soil-plant system; for example, FDE application to pasture must often be supplemented with P and/or K to maximise pasture production.

Excessive application of FDE can result in a range of detrimental effects. Excessive applications of N have been shown to result in N leaching, to levels above the recommended concentration for drinking water (Ledgard et al. 1996b; Ministry of Health 2003); hence most New Zealand regional councils use rules to manage the application of FDE. Councils now tend to specify a N loading rate of between 150–300 kg N ha^{-1} yr^{-1}, down from earlier loading rates of up to 600 kg N ha^{-1} yr^{-1} (Lincoln Environmental 1997). Fertiliser application may have a separate loading rate to animal wastes, but this is true for only a few regions (Close et al. 2001).

While FDE has been recognised as potentially increasing the risk of nitrate-N leaching and microbial contamination and potentially changing soil properties, the current management of FDE is focused on the management of the land application of the N component of FDE (Ministry for the Environment 1999). Recent work on the use of N isotopes (Dittert et al. 1998; Cameron et al. 2003), in a similar manner to that used for groundwater (McLarin et al. 1999), has demonstrated that an N mass balance can be computed and used to assess the importance of the various processes that affect N applied to the soil surface. Nonetheless, water quality management, one of the key drivers behind the move to land application, recognises that changes to water quality are not limited to changes due to nitrate-N leaching (e.g., ANZECC 2000; Ministry for the Environment 2003; Ministry of Health 2003). Land application of FDE has the potential to increase microbial contamination, and hence affect human health (McLeod et al. 2001, 2003; Sinton 2001). An alternative, and important, consideration is that groundwater standards and contamination have important implications on agricultural management and economics (Eco-Link Limited 2000). For example, poor quality water can necessitate costly treatment or the location of alternative supplies; hence, prevention of nitrate contamination rather than remediation is preferred.

The potential negative impacts of FDE on soil properties are probably less than that of other effluent because it is unlikely to contain contaminants such as heavy metals. However, effluent irrigation of largely organic origin has been observed to lead to considerable impacts on soil properties. For example, Degens at al. (2000) noted that, after 22 years of dairy factory effluent application onto a Waikato allophanic soil (Horotiu silt loam), 8% of the N and 91% of the P applied was stored with the soil profile. The effluent irrigation was also associated with increased microbial biomass C and basal respiration. Hence, the effluent irrigation increased the total soil nutrient stores. Since the majority of N in FDE is organic, and so is only slowly available, the application of FDE would increase the availability of soil N over the long term.

Despite the considerable number of studies on FDE, and the land application of effluent generally in New Zealand, the applicability and transferability of results remains problematic for a number of reasons:

1. Many of the studies assessing impacts of FDE application on some of the key soil properties were conducted in the laboratory or using lysimeters. These studies are good for impact assessment in a controlled environment but the transferability of these results is not always possible, as noted by Zaman et al. (1999a,b) and Houlbrooke et al. (2004).
2. The nature of dairy farming in New Zealand has changed considerably. For example, Goold (1980) monitored the effluent from a herd of 102–130 cows on a 53 ha farm. The average dairy farm is now 103 hectares milking 271 cows with a higher stocking density and regular applications of N fertiliser (Wang et al. 2004). As noted by Longhurst et al. (2000), changes in practice can impact on effluent composition (Table 1).
3. The transferability of results conducted with N fertiliser compared to N application via FDE

may be problematic because FDE is not just N, and the addition of a considerable amount of organic C has been observed to change the nature of the soil response to the additions of N. Recent research strongly suggests that the rules relating to N application should consider the source of the N (Zaman et al. 2002; Cameron & Di 2004).

4. Changes to soil properties are slow and gradual and are not always easy to assess. The application of FDE is the application of nutrients in a number of forms that may only become plant-available slowly and may only leach slowly. There are instances in the United States where the capacity of the soil to store nutrients is reached and the soil subsequently acts as a point source (Philips 2002b). Also, the effects of effluent application are not limited to the effects on one property; rather it is the combined effect that is important.
5. The majority of the studies conducted in New Zealand used Waikato (e.g., Te Kowhai and Horotiu) or Canterbury (e.g., Templeton and Lismore) soils reflecting the location of major research centres. While the overall processes and understandings may be transferable, and so models such as the NLE (Di & Cameron 2002b) and OVERSEER™ (Ledgard et al. 1999) developed, a number of important processes, e.g., denitrification and bypass flow, have been shown to vary in their importance as a function of soil type (Strong et al. 1999; Aislabie et al. 2001; McLeod et al. 2001).
6. The methods of analysis and the properties analysed and monitored are variable and have changed over the course of the last 20 years.

Nitrate-N contamination of groundwater, and its associated impacts on surface water, continues to be a problematic and significant concern in New Zealand, for regulators, land managers and waterway users. Hence, management of FDE has emphasised the management of its N component. This emphasis has extended to the development and testing of a dicyandiamide nitrification inhibitor (Di & Cameron 2004). However, while treating grazed pasture with a nitrification inhibitor has been shown to result in substantial environmental benefits (reducing nitrate-N leaching losses), the residual effects of the added nitrification inhibitor and its breakdown products after being added to soils have not been studied in New Zealand soils. This could affect the environmental health and soil microbial diversity. Furthermore, economic benefits (increased nutrient use efficiency and pasture production) and human health impacts of land use and FDE application require a broader appreciation of the components of FDE. FDE application usually appears to lead to increases in the concentration of nutrients such as N and P, organic C and organic matter, and plant-available nutrients such as K, Ca and S. However, the application rate of these may exceed the soil-plant systems' ability to utilise them and so leaching can occur. Over time, the soil's ability to continue to absorb these nutrients may also be exhausted, thus leaching will occur. Also, these changes in soil chemical properties affect, and are affected by, the soil's physical properties; e.g., increases in SOM content can affect infiltration capacity, hydraulic conductivity, and porosity. At present we have a number of tools (e.g., the NLE and OVERSEER™ models), which are able to predict the potential leaching losses associated with N inputs; however, similar tools for predicting the effects on soil properties do not exist. Successful environmental and economic management of dairying in New Zealand, particularly using long-term field-based studies on a range of soil types, requires ongoing research to ensure land application of FDE does not simply result in the transfer of effluent management from a waterway "problem" to a future land "problem".

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