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Effect of treated farm dairy effluent on *E. coli*, phosphorus and nitrogen leaching and greenhouse gas emissions: a field lysimeter study

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Abstract

Purpose Land application of farm dairy effluent (FDE) to pasture soils is the preferred practice in New Zealand. Recently, a new FDE treatment technology has been developed to recycle the water for washing the yard Cameron and Di (J Soils Sediments 2018). Here we report a lysimeter study to compare the leaching losses of *Escherichia coli*, phosphorus (P), and nitrogen (N) and emissions of greenhouse gases from the treated FDE compared with the untreated original FDE.

Materials and methods Lysimeters were collected from a Balmoral silt loam soil (Typic Dystrudept, USDA) and installed in a field trench facility. Treatments included (1) treated effluent (TE), (2) a mixture of TE and recycled water (M), (3) untreated original FDE (FDE), and (4) water as control. The effluents were applied at a surface application rate of 24 mm on each lysimeter in May and again in September 2017. Measurements included leaching losses of *E. coli*, total phosphorus (TP), dissolved reactive phosphorus (DRP), total mineral nitrogen (TN), ammonium-N (NH_4^+-N), and nitrate-N (NO_3^--N); emissions of nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4); herbage yield; and N uptake.

Results and discussion The results showed that *E. coli*, TP, and DRP leaching losses from the TE were 1.31×10^{10} cfu/ha, 0.26 kg P/ha, and 0.009 kg DRP/ha and from M treatments were 6.96×10^{8} cfu/ha, 0.18 kg P/ha, and 0.004 kg DRP/ha, respectively, which were significantly ($P < 0.05$) lower than those from the FDE which were 4.21×10^{10} cfu/ha, 1.75 kg P/ha, and 0.034 kg DRP/ha, respectively. There were no significant differences in NO_3^--N leaching losses amongst the different forms of effluents. There were no significant differences in total N_2O, CO_2 emissions, and CH_4 uptakes from the different effluents ($P < 0.05$). Herbage dry matter yields and N uptakes were also similar in the different effluent-treated lysimeters.

Conclusions Results from this research indicate that land application of the treated effluents (TE) or a mixture of TE plus clarified water (M) would result in significant environmental benefits by reducing *E. coli* and P leaching without increasing greenhouse gas emissions.

Keywords *E. coli* leaching · Farm dairy effluent · Greenhouse gas emissions · Nitrogen leaching · Phosphorus leaching

1 Introduction

The New Zealand dairy industry has grown rapidly over the past three decades, with the number of dairy cows increasing from 2.09 million in 1975/1976 to 4.86 million in 2016/2017 (Dairy 2018). This has led to increased volumes of farm dairy effluent (FDE) being produced on dairy farms. FDE is a mixture of faeces, urine, soil, feed, milk, and water and is produced during the washing of the farm yard and the milking parlour (Toor et al. 2004). As FDE contains valuable plant nutrients such as N, P, K, and S, the application of FDE onto land can improve soil fertility and increase the sustainability of farming systems (Di and Cameron 2002a; Li et al. 2014). Land application of FDE to recycle nutrients

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has become the most common effluent management system in most parts of the world (Wang et al. 2004; Li et al. 2014; Maillard and Angers 2014), particularly in New Zealand (Laubach et al. 2015).

However, there are environmental concerns about land application of FDE because it can be a non-point source of pollution, creating adverse impacts on air and water quality (Cameron and Di 2004; McLeod et al. 2014; Laubach et al. 2015). Government regulations in New Zealand are increasing the pressure on intensive pastoral farming systems, such as dairying, to reduce environmental impacts, including nutrient leaching, harmful bacteria leaching, and greenhouse gas (GHG) emissions (Vogeler et al. 2013). There are also concerns about the amount of freshwater being used on New Zealand dairy farms.

Enhancing water use efficiency and reducing the volume of FDE produced on dairy farms would be an effective way to increase the environmental sustainability and reduce environmental impacts of dairy farming. Therefore, a new FDE treatment technology (commercially known as ClearTech™) has been invented to recycle the water in FDE to wash the farm yard and reduce the volume of FDE applied to the land (Cameron and Di 2018). The treatment technology uses a coagulant to flocculate and settle the colloidal material in FDE and thus produce clarified water (CW) for recycling to wash the yard. This also results in a smaller volume of effluent being produced (i.e. the portion settled to the lower part of the treatment tank) which needs to be irrigated on farm. However, the potential impacts of land application of treated FDE or a mixture of treated effluent and clarified water on nutrient losses and GHG emissions are unknown. The aim of this research was therefore to quantify the nutrient leaching losses and GHG emissions following the application of the treated FDE compared with the original untreated FDE. The objectives of this study were therefore to quantify the P, N, and *Escherichia coli* leaching losses and GHG emissions from the TE and a mixture of TE plus clarified water compared with the untreated FDE in a field lysimeter study.

2 Materials and methods

2.1 Site description and trial setup

The lysimeter study was conducted at Lincoln University, Canterbury, New Zealand (43° 38′ 52″S, 172° 28′ 07″E), where the annual average temperature is 11.5 °C and the annual rainfall is 630 mm.

The soil used was a Balmoral stony silt loam (Typic Dystrudept, USDA) collected from Lincoln University's Ashley Dene Research and Development Station situated near Springston on the Canterbury Plains, New Zealand (NZGD2000: 43° 38′ 42″S, 172° 20′ 33″E). The soil texture in the first 15 cm was a moderately permeable, well-drained silt loam, but the soil profile became increasingly stony below this depth with ~50% of the soil volume occupied by stone by 30-cm depth, and the remaining volume interspersed with fine-to-coarse sands. The soil supported established pasture consisting of perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.), with a minor weed component. The pasture had a history of regular cattle grazing and fertiliser application. The site was fenced off 4 months prior to lysimeter collection to prevent animal access. Soil chemical and physical properties are presented in Table 1.

In March 2017, 20 undisturbed soil monolith lysimeters (0.5-m diameter, 0.7-m depth) were collected using well-established protocols and procedures described in Cameron et al. (1992). The lysimeters were transported to Lincoln University's Field Research Centre next to the Lincoln University campus, and were installed in a purpose-built field trench, with the surface of the lysimeters at the same level as the surrounding field; this ensured that the lysimeters were exposed to the same environmental conditions as the rest of the field. Plastic tubing was connected to the base of each lysimeter and fed into a 10-L container for leachate collection.

2.2 Lysimeter treatments and maintenance

Fresh FDE was collected from Lincoln University dairy farm after milking in the morning of the treatment application day. This farm had Friesian-Jersey milking cows that had grazed outdoors on a ryegrass–white clover mixed pasture. The coagulant (poly ferric sulphate; Cameron and Di 2018) was added to the fresh FDE to flocculate the solids in the effluent. After about 30–60 min, the FDE separated into two layers: (i) upper clarified water (CW) for recycling and (ii) the lower treated effluent (TE) which had a higher concentration of solids. Subsamples of each effluent type were taken for analyses as described in Cameron and Di (2018) (Table 2).

In order to simulate the re-cycling of the clarified water multiple times, an additional 200 mg N/L in the form of ammonium chloride was spiked into the clarified water. This spiked CW was then mixed with the TE in a ratio of 1:3 (1 CW spiked with ammonium chloride to 3 TE volume:volume) to simulate the effluent composition in the dairy pond and represent the material that is actually irrigated on to the pasture in practice.

Four treatments were established in a randomised block design with four replicates. The first treatment was applied on 17 May and the second treatment on 20 September 2017. Three types of effluents: (i) untreated effluent (FDE), (ii) treated effluent (TE), and (iii) a mixture of TE and clarified water that had been recycled (M), were applied at the maximum rate of 24 mm (4.71 L per lysimeter) allowed under local regulations (Table 2). This was equivalent to 222, 226, and 72 kg N/ha for the TE, M, and FDE, respectively. The correct volume

Table 1 Physical and chemical properties of the soil used for the study

Soil analysis[a]	Value	Soil analysis[a]	Value	Soil depth (cm)	Soil bulk density (g/cm^3)	Stone volume (%)	Soil porosity (%)	Soil texture
pH	6.1	CEC	16 $cmol^{+1}$/kg	0–10	1.14	11	58	Stony ZL
Olsen-P	33 μg/mL	Reserve-K	3.6 cmol/kg	10–20	1.57	30	42	Stony ZL
Exch-Ca	8.1 $cmol^{+1}$/kg	Sulphate-S	4 μg/g	20–30	1.90	51	29	Stony SL
Exch-Mg	0.5 $cmol^{+1}$/kg	Organic-C	45 g/kg	30–40	2.05	54	24	Stony S
Exch-K	0.4 $cmol^{+1}$/kg	Total-N	4.0 g/kg	40–60	1.96	51	27	Gravelly S
Exch-Na	0.2 $cmol^{+1}$/kg	Organic-S	5 μg/g					

ZL silt loam, SL sandy loam, S sand

[a] 0–7.5 cm soil

Table 2 Chemical and physical characteristics of the different types of effluents applied to the lysimeters

		Turbidity (NTU)	*E. coli* (cfu/100 mL)	Total-P (g/m^3)	DRP (g/m^3)	Total-N (g/m^3)	NH_4-N (g/m^3)	NO_3-N (g/m^3)	Total-C (g/m^3)
Application 1	untreated FDE	2655	181,333	37	17	131	23	0.10	970
	Treated effluent	4485	27	95	0.05	420	49	6.00	2900
	Mixture of treated clarified water and treated effluent	6914	7	94	0.02	467	101	6	380
Application 2	untreated FDE	2110	1,146,667	26	6.33	170	36	0.12	1270
	Treated effluent	7035	19,300	89	0.16	503	84	4.90	2933
	Mixture of treated clarified water and treated effluent	4969	30,000	65	0.10	473	129	4.77	2600

of effluent was poured directly and uniformly onto the entire surface area of the lysimeters, with the same volume of water applied to the control.

The lysimeters were divided into four blocks, with the four different treatments allocated randomly in each block. Before application, the grass was cut to a uniform height of 50 mm, which is a typical pasture grazing residual height.

Lysimeters were fitted with a rain irrigation simulation system (RISS). Each lysimeter had a spray nozzle (Tee Jet FL-5VC) mounted directly over the top of it. Water was applied to lysimeters either as simulated rain or irrigation. During the winter season (May to September), rainfall was supplemented with randomly simulated rain to the 75th percentile of local winter rainfall records. During the summer season the lysimeters were irrigated at regular rates and time intervals to account for evapotranspiration and prevent soil moisture deficits, following local farming practice. Throughout the experiment, climate data, including rainfall, air, and soil temperatures (10-cm depth), were collected daily from a nearby weather station. The grass was cut periodically to simulate typical grazing practice, and weeds were removed by hand.

2.3 Sample collection

2.3.1 Leachate sampling and analyses

Leachate was collected from the lysimeters on 31 occasions following irrigation and/or significant rainfall that occurred from May 2017 to May 2018. Leachate volume was recorded and subsamples were retained for analyses of NO_3^--N, NH_4^+-N, TP, DRP, and *E. coli* (as required). NO_3^--N and NH_4^+-N were analysed by FIA (Tecator Inc., Sweden), DRP was directly measured on filtered (0.45 μm) leachate samples by the malachite green method (Ohno and Zibilske 1991), and TP was determined by digesting samples with $K_2S_2O_8$ and NaOH (Ebina et al. 1983) followed by the malachite green colorimetry method. The numbers of *E. coli* in 100-mL aliquots were determined using MPN count using Colilert (incubated at 35 °C for 24 h), or 1–20 Colilert 18 (incubated at 35 °C for 18 h) (as described in Cameron and Di 2018). Nitrogen, phosphorus, and *E. coli* losses from the lysimeters were calculated by multiplying the volume of leachate with the concentrations at each sampling time. Total mineral N leaching loss represents the combined leaching losses of NO_3^--N and NH_4^+-N.

2.3.2 Gas sampling and analyses

GHG collection and measurement were carried out the day following treatment application and thereafter twice a week for the first 2 months and then once per week until background levels were reached. On each sampling day, gas samples were collected between 1 pm and 2 pm during which time the air temperature represented the daily average, and the GHG emissions were similar to the daily average (Di et al. 2007). A closed chamber method similar to that described by Hutchinson and Mosier (1981) was used to determine GHG emissions from the treated lysimeters. The gas chamber was constructed of a metal cylinder insulated with 2.5-cm-thick polystyrene foam to avoid heating of the atmosphere in the chamber during sampling. During periods of GHG measurement, the chamber was placed inside a small water trough which was mounted around the top of each lysimeter casing for gas sampling. At each sampling time, the chamber was placed on top of the lysimeters for a total of 40 min, and three samples, 20 min apart, were taken using a syringe through a rubber septum on top of the gas chamber.

GHG concentration analyses were conducted by gas chromatograph (GC) (Model 8610C, SRI Instruments, CA, USA) with an automated Gilson GX-271 auto sampler (Gilson Inc., MI, USA) coupled to an electron capture detector (ECD) and flame ionised detector (FID). N_2O was measured by ECD, and CH_4 and CO_2 were measured by FID (CO_2 was converted into CH_4 by methanizer and then measured). The GC used three HayeSep D packed column as the precolumn, and two of six HayeSep D as the analytical column. In terms of gas supply, the oxygen free nitrogen and 10% methane in Ar were used as carrier gas and ECD make-up gas, respectively. H_2 and air were used for FID flame. For the detector temperature setting, ECD was at 310 °C and FID was at 370 °C.

Hourly GHG emissions were calculated based on the rate of increase in GHG concentration in the chamber (from 0 to 40 min) corrected for temperature and the ratio of surface area to chamber volume. Daily emissions were calculated using the hourly flux, assuming that it represented the average hourly flux of the day (Hutchinson and Mosier 1981; Rochette 2011). Cumulative emissions were calculated by integrating the calculated daily GHG fluxes and linear interpolation between measurement points. The N_2O emission factor (EF_1) was calculated for each effluent N rate as

$$EF_1\,(\%) = 100 \times (N_2O{-}N\ \text{treatment}{-}N_2O{-}N\ \text{water})/\text{effluent}{-}N\ (\text{applied}).$$

2.3.3 Herbage sampling

The grass was cut when it was 200 mm high using electric grass shears to simulate typical grazing practice. All the harvested herbage was removed and dry matter yield recorded. Herbage total N was analysed on an Elementar Vario-Max CN Elemental Analyser (Elementar GmbH, Hanau, Germany).

2.3.4 Data analysis

The treatment effects on TP, DRP, NO_3^--N, NH_4^+-N, and *E. coli* leaching losses and GHG emissions were statistically analysed by conducting an analysis of variance (ANOVA) using GenStat (14th edition, Lawes Agricultural Trust) to test for treatment effect significance.

3 Results

3.1 Climate conditions

Cumulative water inputs during the experimental period were 1504 mm (Fig. 1), of which 521 mm came from irrigation. The maximum monthly rainfall (155.6 mm) occurred in June, which was about 1.6 times higher than the average value for the region. There was very low rainfall in November (1.2 mm) and the first half of December, which was unusual for the area. Therefore, irrigation was applied during summer months to compensate for the evapotranspiration losses. Thirty-one drainage events occurred during the experimental period. The drainage pattern from the lysimeters was consistent with the water inputs; approximately 30% of the applied water was discharged in drainage in all treatments. About 54–57% of the drainage occurred after the first treatment application, with cumulative drainage varying between 459 and 508 mm for different treatments (data not shown). Over the experimental period, the average daily air temperature ranged from a low of 1.9 °C on 30 July to a high of 23.4 °C on 10 December. Daily ground temperature (10-cm depth) ranged from 3.9 °C on 31 July to 24.2 °C on 11 December (Fig. 1).

3.2 Effect of effluents on *E. coli* leaching losses

After the first treatment, *E. coli* breakthrough curves for the FDE, TE, and M peaked at 2176, 548, and 21 cfu/100 mL, respectively, which occurred between 10- and 20-mm drainage (Fig. 2). After the second treatment, the *E. coli* breakthrough curves for the FDE and TE peaked at 2022 and 542 cfu/100 mL, respectively, very similar to the results after the first application. The peak *E. coli* numbers in the leachate in the untreated FDE treatment were significantly higher than those from the TE or M treatments ($P < 0.05$). The *E. coli* numbers declined rapidly to much lower levels after both treatment applications (Fig. 2).

A total of 4.21×10^{10} cfu/ha *E. coli* was leached from the FDE, which was significantly higher ($P < 0.05$) than those in the other treatments during the drainage period (Table 3). There were no significant differences between the other effluent treatments or between the effluent and the control in the amount of *E. coli* leached.

3.3 Effect of effluents on phosphorus leaching losses

Following both effluent application events, the TP concentrations increased significantly in the FDE treatment, reaching peak concentrations of 1.99 and 2.5 mg P/L following the first and second effluent treatments, respectively (Fig. 3a). These peak concentrations were well above those in the TE and the M treatments which had a total P concentration below 0.35 mg P/L (Fig. 3a). The TP leaching loss during the experimental period from the FDE treatment (1.75 kg P/ha) was significantly higher than those from the other treatments (0.18–0.28 kg P/ha) ($P < 0.05$) (Table 3). There was no significant difference amongst the TE, M, and the control treatments in TP leaching losses.

The DRP leaching breakthrough curves also showed higher peak concentrations from the FDE treatment than the other effluent treatments, particularly after the second effluent treatment application (Fig. 3b). Single peaks emerged following the first effluent application but two breakthrough peaks emerged following the second effluent application in the FDE treatments, although there was high variability associated with the concentrations. The DRP concentrations were much lower than the TP concentrations in the leachate. The DRP leaching loss during the experimental period for the FDE treatment was significantly higher at 0.034 kg P/ha than those from the other treatments which ranged from 0.004 to 0.009 kg P/ha. No significant differences were observed in the DRP leaching losses amongst the TE, M, and the control treatments ($P < 0.05$) (Table 3).

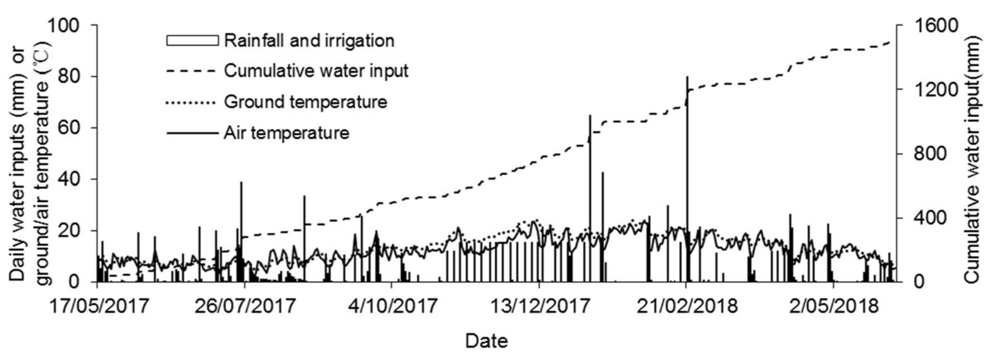

Fig. 1 Daily and cumulative rainfall and irrigation (including simulated rainfall) water inputs, and daily average air and ground temperature for the experimental period

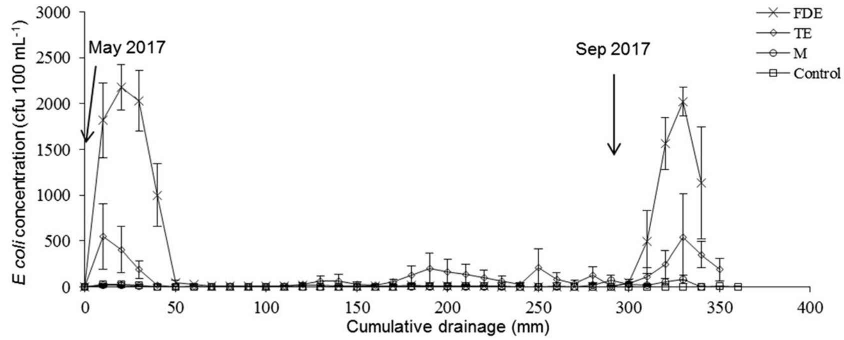

Fig. 2 Average leachate *E. coli* concentrations from the lysimeters

3.4 Effect of effluents on nitrogen leaching losses

Small NO_3^--N concentration peaks from the M (7.90 mg N/L) and TE (5.16 mg N/L) treatments occurred at 10-mm drainage, and at 80-mm drainage, respectively, following the first effluent application. No NO_3^--N concentration peaks were observed in all the treatments following the second treatment application. There were no significant differences in NO_3^--N concentration amongst the different treatments following both treatment applications (Fig. 4).

Approximately 94.2–96.4% of total mineral N losses were as NO_3^--N for the TE, M, and control treatments, whilst a lower proportion of 68.3% for the FDE treatment. The NH_4^+-N leaching loss from the FDE treatment was 0.99 kg N/ha, which was higher than the 0.16–0.28 kg N/ha from the other treatments ($P < 0.05$). There were no significant differences in total NO_3^--N and total mineral N leaching losses amongst all the treatments ($P > 0.05$) (Table 3).

3.5 Effect of effluents on GHG emissions

Daily N_2O, CO_2, and CH_4 fluxes showed no significant differences amongst the four treatments during the experimental period. Therefore, N_2O fluxes are used as an example to show the daily GHG flux variations with time (Fig. 5). Increased N_2O emissions relative to background fluxes were observed the day after the treatment applications. Daily N_2O fluxes showed similar patterns following both treatment applications, with initial high N_2O emissions following effluent applications, which then declined rapidly with time, reaching background levels. Total N_2O emissions over the measurement period ranged from 0.18 to 0.61 kg N_2O-N/ha, but there was no statistically significant difference amongst the different effluent treatments (Table 4). The N_2O emission factors ranged from 0.11 to 0.36%, and again, there were no significant differences amongst the different effluent treatments (Table 4).

Daily CO_2 fluxes from all treatments fluctuated with time, and showed an increasing trend as air and ground temperatures increased (data not shown). Although total CO_2 emission from the M treatment (14,025 kg CO_2-C/ha) was significantly higher than that from the control (12,223 kg CO_2-C/ha) ($P < 0.05$), there were no significant differences amongst the different effluent treatments ($P < 0.05$) (Table 4).

There were no uniform trends in daily CH_4 fluxes over time (data not shown). The soil acted as a sink for CH_4 as shown by the negative CH_4 emission values in Table 4. Total CH_4 uptakes varied amongst treatments, ranging from 0.57 kg CH_4-C/ha in the FDE to 0.16 kg CH_4-C/ha in the control treatment, but there were no significant differences amongst all the treatments (Table 4).

3.6 Effect of effluents on herbage yields and N uptakes

Herbage dry matter yields ranged from 5.7 t/ha in the FDE treatment to 7.7 t/ha in the M treatment (Fig. 6), but there were

Table 3 *E. coli*, P, and N leaching losses over the experimental period

	E. coli (cfu/ha)	P loss (kg P/ha)		N loss (kg N/ha)		
		Total-P	DRP	NO_3^--N	NH_4^+-N	Total-N
FDE	4.21E+10 a	1.75 a	0.034 a	2.14 a	0.99 a	3.13 a
TE	1.31E+10 b	0.26 b	0.009 b	5.92 a	0.22 b	6.14 a
M	6.96E+08 b	0.18 b	0.004 b	7.31 a	0.28 b	7.59 a
Control	7.05E+08 b	0.28 b	0.009 b	2.67 a	0.16 b	2.83 a

Means with different letters within the same column indicates significant difference ($P < 0.05$)

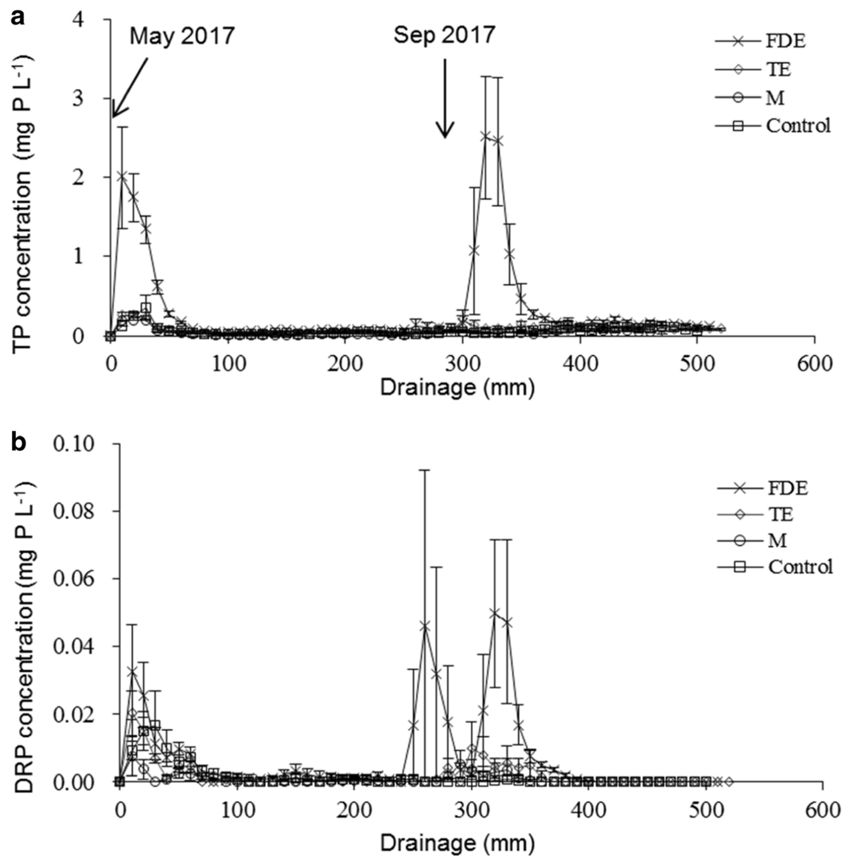

Fig. 3 Average leachate TP and DRP concentrations from the lysimeters

no significant differences amongst the different treatments. Similarly, herbage N uptakes ranged from 142.5 kg N/ha in the FDE treatment to 184.5 kg N/ha in the M treatment (Fig. 6), but there were no significant differences amongst the different treatments. The TE and M treatments showed a trend of 33.6% and 33.8% higher herbage yields and 28.2% and 29.4% higher N uptakes, respectively, compared to the FDE treatment, but these differences were not statistically significant ($P > 0.05$) (Fig. 6).

4 Discussion

Results from this field lysimeter study clearly showed that the land application of the treated effluent (TE) or the TE plus recycled clarified water (M treatment) can lead to significant reductions in *E. coli* leaching, total phosphorus (TP), and dissolved reactive phosphorus (DRP) leaching, compared with the untreated original FDE. The reductions in *E. coli* leaching were equivalent to 69% and 98% for the TE and M treatments,

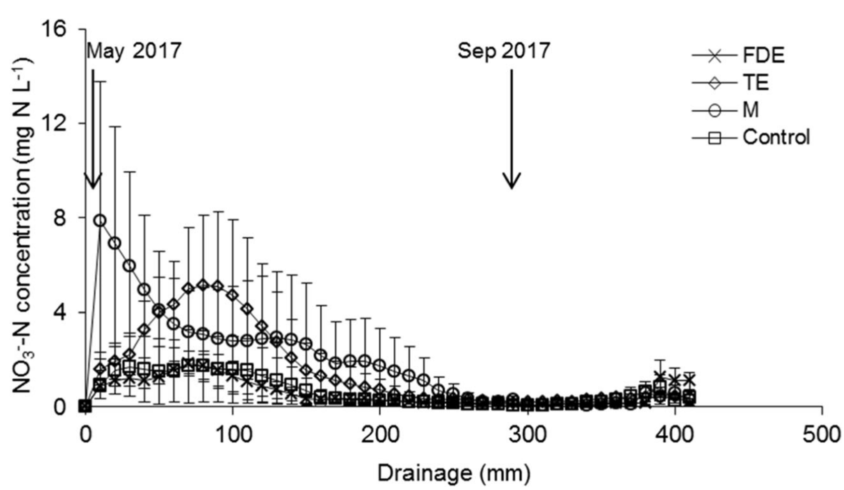

Fig. 4 Average leachate NO_3^--N concentrations from the lysimeters

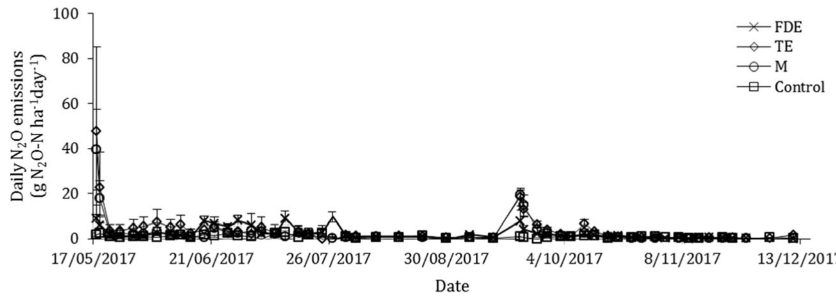

Fig. 5 Daily N_2O fluxes as affected by the application of the different types of effluents

respectively. The reductions in leaching losses were equivalent to 85% and 90% for TP, 75% and 88% for DRP in the TE and M treatments, respectively. These reductions in *E. coli*, TP, and DRP leaching losses represent significant environmental benefits that would be gained by treating the FDE using the latest effluent treatment technology before land application (Cameron and Di 2018).

The significant reductions in *E. coli* leaching loss in the TE and M treatments were attributed to the significant reductions in *E. coli* numbers in these effluents following the treatment of FDE (Table 2). The ClearTech™ effluent treatment technology effectively kills and therefore reduces *E. coli* numbers to very low levels, not only in the clarified water for recycling but also in the TE which contains most of the solid materials that settles at the bottom of the treatment tank (Cameron and Di 2018).

Similarly, the reduction in DRP leaching in the TE and M treatments compared with the original FDE was also due to the significant reduction in DRP concentrations in these treated effluents (Table 2). The coagulant added to treat the FDE contains ferric iron (Fe^{3+}) (Cameron and Di 2018); therefore, a significant proportion of the P in the FDE is transformed to less mobile iron phosphate compounds. This is also why the TP leaching losses were lower from all the treated effluents, despite the TE and the M effluents containing higher TP concentrations (Table 2). The TP and DRP leaching losses from the FDE recorded here were in similar orders of magnitudes to those reported previously (Toor et al. 2004, 2010).

The early breakthroughs of *E. coli*, TP, and DRP straight after the effluent applications would indicate preferential flow through macro-pores (Figs. 2 and 3). This was a reflection of the stony nature of the soil used for the study (Table 1).

Despite the higher N application rates in the TE and M treatments (222, 226 kg N/ha, respectively) compared with the FDE (72 kg N/ha), there were no statistically significant differences in the total NO_3^--N or total mineral N leaching losses amongst the different effluents (Table 3). The largest form of N leached was NO_3^--N, and this agrees with previous studies that have shown that most of the N leached through soil following land application of FDE to soil is NO_3^--N, and only very low amounts of NH_4^+-N leaching losses are detected (Di et al. 1998a, b, 1999; Di and Cameron 2002a, b). The total mineral N leaching losses recorded from the different effluents were very low compared to the NO_3^--N leaching losses recorded under urine patches in grazed pastures which make up the dominant N leaching loss in grazed pastures (Di and Cameron 2002a, b; Dai et al. 2013).

The similar emissions of N_2O and CO_2 and similar CH_4 uptakes in the TE and M treatments as compared with the application of the untreated FDE (Table 4) would indicate that the land application of the treated effluents would not increase GHG emissions compared with the application of the original FDE. The emission factors (EF_1), ranging from 0.11 to 0.36%, were in a similar range to the EF_1 value used in the 2018 New Zealand GHG inventory calculations for FDE (Ministry for the Environment 2018). The large initial N_2O emissions were similar to those reported in some other studies that also showed large N_2O emissions shortly after animal effluent application (Li et al. 2015; van der Weerden et al. 2016). More studies are needed to quantify GHG emissions following the application of the treated effluents to different soils and under different environmental conditions.

The similar herbage yield and N uptakes between the different effluents suggest similar agronomic values of the

Table 4 GHG emissions from lysimeters affected by the application of different types of effluents

	N_2O emissions		CO_2 emissions ($kg CO_2$-C/ha)	CH_4 emissions ($kg CH_4$-C/ha)
	($kg N_2O$-N/ha)	Emission factors		
FDE	0.44 a	0.36% a	12,817 ab	− 0.57 a
TE	0.61 a	0.19% a	13,046 ab	− 0.22 a
M	0.44 a	0.11% a	14,025 a	− 0.29 a
Control	0.18 a	–	12,223 b	− 0.16 a

Means with different letters with in the same column indicates significant difference ($P < 0.05$)

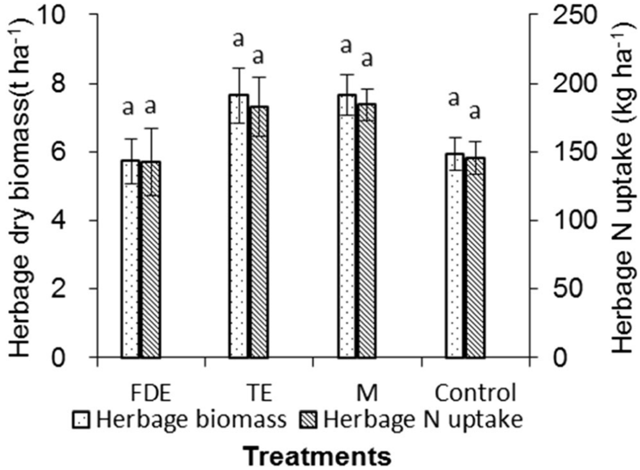

Fig. 6 Herbage dry matter yields and N uptakes as affected by the application of the different types of effluents

treated effluents to the untreated FDE. The 34% higher herbage yields in the TE and M treatments compared with the FDE treatment were not statistically significant.

5 Conclusions

Results from this study indicate that the land application of the treated effluents (TE and M) would result in significant environmental benefits by significantly decreasing *E. coli*, TP, and DRP leaching losses compared with the untreated FDE. The land application of the treated effluents did not result in significantly different N leaching loss or GHG emissions compared with the FDE.

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