



Farm-scale carbon and nitrogen fluxes in pastoral dairy production systems using different nitrogen fertilizer regimes

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Abstract The nitrogen (N) fertilizer application rate ($\text{kg ha}^{-1} \text{ year}^{-1}$) in pastoral dairy systems affects the flow of N through the soil, plant and animal pools of the system. With better understanding of the magnitude of these pools and their fluxes, dairy systems could be managed to improve N use efficiency, therefore reducing losses to the environment. A study with three levels of N fertilizer, 0 (N0), 150 (N150) and 300 (N300) $\text{kg N ha}^{-1} \text{ year}^{-1}$, was conducted in the Canterbury region of New Zealand from 1 June 2017 till 31 May 2018. Farm measurements, e.g. pasture and milk production, were used to calibrate three different farm-scale models, DairyNZ's Whole Farm Model, DairyMod, and Overseer[®]. The models were used to extrapolate periodic farm measurements

to predictions of carbon (C) and N pools and fluxes on an annual basis. Pasture and milk production per hectare increased from N0 to N300 by 70 and 58%, respectively. There was a concomitant increase in farm-gate N surplus (input–output) of 43%, resulting in predicted increases in N leaching and greenhouse gas emissions of 72 and 67%, respectively. By increasing N fertilizer from 0 to 300 $\text{kg N ha}^{-1} \text{ year}^{-1}$, 53% more feed N flowed through the dairy herd with surplus N deposited as urinary N increasing by 49%. Plant uptake and soil immobilization increased by 58 and 343%, respectively, but not enough to avoid substantial increases in leaching and emission losses. Carbon flux through the soil system increased through increased litter and faecal deposition, but with very little C sequestration because of accelerated microbial respiration rates.

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Keywords Modelling · Nitrogen leaching · Greenhouse gas · Nutrient cycling · Animal excreta

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Introduction

The New Zealand pastoral sector relies on the biological processes occurring in soils, forage plants and ruminant animals to drive the production of dairy, meat and fibre products. Microbial communities associated with soils, plants and ruminant animals

use and modify the form and flow of carbon (C) and nitrogen (N) in these environments. Understanding the connections and interactions between these agricultural microbiomes and C and N fluxes will foster new approaches to drive greater agricultural productivity while reducing the environmental impacts of farming ruminant animals (Brussaard et al. 2007).

Pastoral systems involve a strong coupling of C and N through pasture growth and heterotrophic microbial production, but also an uncoupling of the C–N cycles when grazing ruminants release digested C into the atmosphere as respired CO₂ and enteric methane (the greenhouse gas CH₄), and by returning digested N at a high concentration in urine patches (Soussana and Lemaire 2014). In these systems, urine patches are the most important source of nitrate (NO₃) leached to water bodies and nitrous oxide (the greenhouse gas N₂O) emissions to the atmosphere (de Klein et al. 2010). Increasing N fertilizer input is a relatively inexpensive way of intensifying these systems by increasing herbage yield, followed by an increase in animal intake, which may or may not involve increasing animal numbers (cows ha⁻¹). However, there is a trade-off. With intensification there is an increase in C–N coupling, but this is balanced by an increase in uncoupling by the animals (Parsons et al. 2013). Stimulation of vegetation growth by abundant N and phosphorous (P) fertilizer applications increases the C flows from the atmosphere to the soil, while increasing animal intake reduces mean C residence time within the system (Soussana and Lemaire 2014), and converts more N to forms readily lost to the environment.

It is well known that N fertilizer use is invaluable for maintaining agricultural production, but its use, and particularly inefficient use, can lead to environmental losses (de Klein et al. 2017). The question is at what point does N fertilizer input becomes inefficient i.e., where does the balance swing from being favourable for production (coupling) to being unfavourable for the environment (uncoupling)? For instance, Ledgard et al. (1997) showed in a pastoral dairy trial in New Zealand that increasing N inputs (fertilizer, imported feed, N fixation) increased N surpluses ($N_{\text{input}} - N_{\text{output in product}}$) and led to relatively large increases in gaseous losses and nitrate leaching. Applying N fertilizer at 400 kg N ha⁻¹ - year⁻¹ resulted in a low N use efficiency ($N_{\text{output}}/N_{\text{input}} = 28\%$) and had an adverse effect on

groundwater nitrate levels. In contrast, the most efficient dairy system used no N fertilizer, had an N surplus of only 100 kg N ha⁻¹ year⁻¹, and resulted in very high (45%) conversion of N input to product. However, the last-mentioned system was clearly not optimum for food production, so probably not striking the right balance between production and the environment.

The main effect of fertilizer N use on N losses from grazed pastures is indirect, with higher fertilizer N inputs increasing pasture production and animal intake, and thus excretal losses (de Klein et al. 2010). This was confirmed by Parsons et al. (2016) in a modelling study where they focussed on intensification of a pastoral dairy system from increases in N inputs via fertilizer and supplements, and the outcomes of this in the longer-term sustainable balance of production versus environmental impacts. They confirmed that the main driver of N release (N leaching and N₂O emissions) are altered inputs of N and, less so, animal numbers in response to these. The importance of N input in the balance between food production (milk in this study) and environmental impacts was further shown by Chapman and Parsons (2017) who used modelling to demonstrate the trade-offs between food production and the long-term environmental impact of meat and dairy production systems in New Zealand. They showed that it is not the change in land use from meat animals to dairy animals per se that has led to greater environmental impacts of pastoral systems. Rather, it is the level of N inputs associated with intensification of dairying that matters. Their analysis showed that land use change from meat to dairy (with total N inputs of around 150 kg N ha⁻¹ year⁻¹ in dairy) can result in a balance favouring food production with a relatively low environmental impact per hectare.

The role of synthetic N fertilizer input in soil C storage is another key consideration in these pastoral systems. The assertion has been made that this N input increases soil organic C by enhancing the production of vegetation residues. Yet the opposite effect was reported long before the modern era of chemical based N management (Albrecht 1938), which is fully consistent with evidence that mineral N enhances microbial decomposition of plant residues (Khan et al. 2007). Given the fundamental coupling of microbial C and N cycling, the dominant occurrence of both elements in soil organic forms, and the close

correlation between soil C and N mineralization, the loss of soil organic C has serious implications for the storage of soil N (Mulvaney et al. 2009).

Nitrogen input into pastoral dairy systems has significant ripple effects through the whole system (e.g. C and N fluxes) that needs better understanding and quantification to find the balance between food production and environmental impact. This paper forms part of a larger study with the overall objective of looking for associations between soil, plant and animal microbiome characteristics and C and N fluxes under different N fertilizer inputs to a pastoral dairy system. Here we report on the use of models to estimate the magnitude of the effects of different N fertilizer inputs on farm-scale C and N pools and fluxes, with the specific objective of quantifying N output in product relative to environmental losses. We hypothesised that an increase in N fertilizer would increase N output, but also the surplus in the system, and consequently N losses to the environment. Furthermore, we anticipated an increase in the amount and rate of C cycling through the soil pools.

Materials and methods

Farm measurements and lysimeter study

An experiment was designed to evaluate the effect of three rates of N fertilizer application on perennial ryegrass-white clover (*Lolium perenne* L.–*Trifolium repens* L.) swards at Lincoln University's Ashley Dene Research and Development Station (ADRDS), Canterbury, New Zealand (− 43.6468, 172.3467). During the farm season of 1 June 2017 to 31 May 2018 urea, as a source of N, was applied to established perennial ryegrass-white clover swards to achieve rates of 0, 150 and 300 kg N ha^{−1} year^{−1}. Urea was applied at approximately 60 kg (~ 28 kg N ha^{−1}) or 120 kg (~ 55 kg N ha^{−1}) per dressing 10 days after grazing for treatments 150 and 300 kg N ha^{−1} year^{−1}. The levels of N addition were chosen as they span the range of N application rates relevant to pastoral irrigated Canterbury dairy farms. The 300 kg N ha^{−1} year^{−1} treatment (N300) represents the high-end input, while 150 kg N ha^{−1} year^{−1} (N150) represents the low end. The zero urea treatment (N0) would not be expected on a conventional Canterbury dairy farm, but was included as an

experimental condition which allowed assessment of soil, plant and animal microbiomes under potentially N-deficient conditions where endogenous N fixation via clover and dairy effluent spreading were the only sources of background N. The swards allocated to this experiment were grazed by lactating dairy cows as part of the normal rotational grazing practice of the larger experimental farm. When grazing was due, cows were allocated the same herbage allowance (approximately 30 kg DM cow^{−1} day^{−1} to ground level) across the three treatments by adjusting the size of the allocated break. This was done to maintain consistent herbage intakes per cow and grazing pressures across treatments. Ten lactating dairy cows were grazed in two groups of five in each of the experimental swards for 2 weeks prior to sampling periods. The 10-day sampling periods were chosen for November 2017 (spring, ~ 110 days in milk, DIM), and in April–May 2018 (autumn, ~ 260 DIM). The November 2017 time point enabled sample collection during maximum herbage growth and milk production, while the April–May 2018 sampling coincided with declining herbage growth and nutritive value and with declining milk output prior to drying off in May 2018. During the sampling periods milk yield was measured with in-line milk meters in the milking parlour and composition determined using a MilkoScan machine (Foss MilkoScan FT1, MilkTestNZ, Hamilton). Liveweight was determined using walk-over scales as cows left the milking parlour after each milking.

Swards allocated to this experiment were examined weekly with a rising-plate meter (RPM) for pasture accumulation estimates, and daily during the 10-day sampling periods to determine daily feed allocation. The readings were converted to average herbage mass using the equation $\text{RPM} \times 140 + 500 \text{ kg DM/ha}$ (DairyNZ 2017). Pre- and post-grazing readings were used to estimate DM intake (DMI) and herbage accumulated since the last grazing event. Herbage samples were collected monthly and sub-sampled for DM% and chemical composition. The samples for chemical composition were freeze-dried, ground and analysed for NDF, ADF, WSC, CP, and ME using near infrared and wet chemistry (Riddolls laboratory, Lincoln University, Christchurch).

To assess N loss in leachates from soils under the treated swards, eight lysimeters on each of the N0, N150 and N300 treatments were installed in purpose-built lysimeter facilities within adjacent paddocks of

each treatment at ADRDS. Undisturbed soil monolith lysimeters, 50 cm diameter and 70 cm deep, were sampled following well-established protocols and procedures (Cameron et al. 1992). To account for variations in cow urine deposition onto pastures, 4 lysimeters from each treatment received cow urine, while 4 remained without urine. The urine was collected from the cows grazing the experimental swards and applied on 2 May 2018 to represent the typical effects of urine deposition on the trial sites during autumn grazing. The lysimeters received the same water and fertilizer (8 applications, after grazing, at 21–35 day intervals, with no applications during the winter months) inputs as the soils in experimental swards. Pasture on the lysimeters was grazed at the same time as the main area of the experimental sward was grazed. The leachates from the lysimeters were collected when drainage was greater than 200 ml or weekly (as required), and were analysed using FOSS FIAstar 5000 twin channel analyser to determine nitrate, and ammonium concentrations.

Models and simulation study

Since production data from the experiment were only collected at certain time points, November 2017 (spring) and April–May 2018 (autumn), farm-scale models were used to predict the bio-physical performance over a full farm season (1 June 2017–31 May 2018). Three models were used to describe the different components of the system. The DairyNZ Whole Farm Model (WFM; Beukes et al. 2008) was used for calibration against pasture and animal production data collected during the 10-day sampling periods and for extrapolating these data to production over a full season. The WFM was also interrogated for C and N pools and fluxes. DairyMod (Johnson et al. 2008) was used for C and N pools and fluxes, and Overseer[®] (Wheeler et al. 2003) as a long-term average model for N leaching and greenhouse gas (GHG) emissions. See the Online Resource for a brief description of the key elements of each model (Supplementary Material).

The WFM was initialised for the three treatments by starting with N300 and running it with daily climate data from the Lincoln weather station (−43.625S, 172.475E) (data supplied by National Institute of Water and Atmospheric Research), approximately 5 km from ADRDS. The WFM feed composition

table for Canterbury ryegrass-clover pasture was adjusted for the spring months (Sep–Nov) and the autumn months (Mar–May) to reflect the measured pasture quality from the experiment for the November-2017 and April–May-2018 sampling points (Table 1). The scenario was run over two consecutive years with the first year (2016–2017) used as a run-in year for setting up pasture covers and soil conditions for the second year. Output was collected for the second year (2017–2018) only. Observed and predicted annual pasture yield for 2017–2018 farm season was used to determine a stocking rate of 3.6 cows ha^{−1} (a comparative stocking rate of approximately 95 kg live weight per t DM feed offered) for N300, which is in the expected range for irrigated Canterbury dairy systems (LIC and DairyNZ 2017). In the model, feed deficits for lactating cows were filled with home-grown plus imported pasture silage. Non-lactating cows were sent off farm for the winter months (June–July) and were not included in any of the model outputs. This scenario was modified by reducing the N fertilizer to 150 and 0 kg ha^{−1} for the N150 and N0 scenarios, respectively. Feasible scenarios were achieved by reducing the stocking rate according to the annual pasture yield to maintain a comparative stocking rate equivalent to the N300 scenario (~95 kg live weight per t DM feed offered). The WFM generated output on monthly and annual pasture growth, pasture and supplements eaten, annual milk production per cow and per hectare, N balance per animal and per hectare, and GHG emissions. Daily urination events, N fertilizer and irrigation applications were used in the WFM–urine patch framework–APSIM model combination, together with the Lincoln climate file, to predict N leaching below the fourth layer of the Lismore/Balmoral soil (55 cm) (see Beukes et al. 2011 for more detail on this procedure). The WFM was also interrogated for predictions of pools and fluxes of C and N in the first soil layer (topsoil of approximately 20 cm).

DairyMod (version 5.7.5) was initialized for the three systems, N0, N150 and N300, following the same stocking rate as determined with the WFM. The model was run with the same Lincoln climate file as used in the WFM but pre-running required more climate years. The model was run with Lincoln climate from 1991 through two loops (2 × 1991–2018) giving 53 run-in years before taking the output for the last year to ensure soil pools and

Table 1 Pasture chemical composition for spring (November 2017) and autumn (April–May 2018) ryegrass-white clover pastures at Ashley Dene Research and Development Station with different levels of N fertilizer input, 0, 150 or 300 kg N ha⁻¹ year⁻¹

N input	Season	Dry Matter %	ADF %DM	WSC %DM	NDF %DM	CP %DM	ME MJ kg DM ⁻¹
0	Spring	22.4 ^a	22.0 ^a	21.5 ^a	40.2 ^a	14.8 ^a	12.8 ^a
	Autumn	13.7 ^b	24.0 ^b	9.7 ^b	42.4 ^b	23.9 ^b	12.05 ^b
150	Spring	21.9 ^a	22.3 ^a	22.6 ^a	38.5 ^a	15.3 ^a	12.75 ^a
	Autumn	13.7 ^b	23.3 ^b	7.0 ^b	41.4 ^b	28.1 ^b	12.15 ^b
300	Spring	24.5 ^a	21.4 ^a	21.0 ^a	40.0 ^a	16.3 ^a	12.55 ^a
	Autumn	12.8 ^b	23.9 ^b	6.9 ^b	42.3 ^b	28.3 ^b	12.0 ^b
	SEM	0.56	0.70	2.01	1.60	1.14	0.14

Numbers followed by different superscript letters are statistically different in the same column ($P < 0.05$)

ADF acid detergent fibre, *WSC* water soluble carbohydrate, *NDF* neutral detergent fibre, *CP* crude protein, *ME* metabolizable energy

fluxes have stabilized. The soil was initialized with the default hydraulic properties for a light soil, with low organic matter content. Paddocks were initialized with perennial ryegrass—white clover mix but certain parameters had to be changed from the default to achieve approximately the same annual pasture yield as predicted with the WFM. For perennial ryegrass the root depth was changed from 40 to 80 cm, depth for 50% of the root distribution from 15 to 30 cm, the scale factor between wilting point and field capacity for onset of water stress from 0.8 to 0.4, and transpiration reduction at saturation from 0.5 to 1.0. For white clover the root depth was changed from 40 to 60 cm, and depth for 50% of root distribution from 15 to 30 cm. Default settings for dairy cows were changed from normal Friesian live weight of 600 kg per animal to 480 kg for the smaller Friesian-Jersey cross-bred cows. Milk protein proportion was changed from 3 to 3.2% and efficiency of milk production from 60 to 55%. Management policies followed best management practice for conserving surplus pasture, stock rotations, silage feeding, and N fertilizer and irrigation applications. Model output was collected on C and N pools and fluxes in soil, plant, animal and atmosphere for the last year of the multiple-year simulations.

Overseer[®] version 6.3 was set up for each of the three systems using averages for the Lincoln climate, default settings for a Lismore soil, cross-bred cows, with stocking rate, milk production and imported pasture silage as predicted by the WFM. Output collected was on N pools and fluxes, and GHG

emissions. There is no single, country-wide regulatory limit on N leaching from dairy farms in New Zealand. Limit setting is delegated to regional and district authorities. Typical leaching losses from irrigated dairy pastures on this free-draining, stony soil type is approximately 65 kg N ha⁻¹ year⁻¹, which farmers will have to reduce by 30% by 2022 (Environment Canterbury, Unpublished Plan Change Notice, June 2016).

Data analyses and presentation

During the two sampling periods of the trial, 11–21 November 2017 (spring) and 21 April–1 May 2018 (autumn) milk production per cow was measured every day. Treatment averages were calculated and plotted against date. As part of ground-truthing the model, the WFM predicted milk production per cow was extracted and plotted against the observed values for the same dates. Since the WFM produced output for the full farm season 1 June 2017 to 31 May 2018, predicted data points were plotted for every day from 11 Nov 2017 to 1 May 2018. The R software package was used to plot average milk production per cow per treatment as smoothed lines against date by fitting a polynomial surface with error margins using the local regression (loess) function in R.

Results

The animal trial and simulation study

The measured pasture chemical composition from the trial for the November-2017 and April–May-2018 sampling points are reported in Table 1. The compositions did not differ between the N treatments. Differences between seasons were significant ($P < 0.05$) with no N treatment \times season

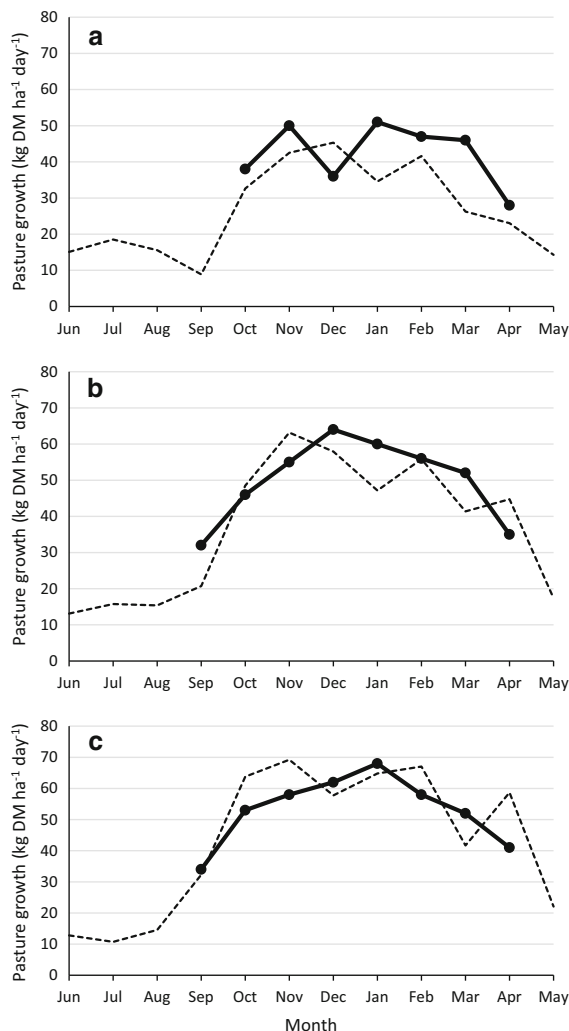


Fig. 1 Observed (solid line) versus predicted (dashed line), using the DairyNZ Whole Farm Model, pasture growth rates ($\text{kg DM ha}^{-1} \text{ day}^{-1}$) for the Ashley Dene pastures receiving either 0 (a), 150 (b) or 300 (c) $\text{kg N fertilizer ha}^{-1}$ for the 2017–2018 farm season. Modelling used actual climate data from the Lincoln weather station for 2017–2018

interactions. The clover content of the pastures varied from 9 to 24% depending on the season.

The predicted pasture growth rates generally followed the trends of the observed with peak growth rates increasing from N0 to N150 to N300 by about $10 \text{ kg DM ha}^{-1} \text{ day}^{-1}$ (Fig. 1). The predicted annual pasture yield increased by $3600 \text{ kg DM ha}^{-1}$ when N fertilizer increased from zero to $150 \text{ kg N ha}^{-1} \text{ year}^{-1}$, with a further increase of $3100 \text{ kg DM ha}^{-1}$ when fertilizer was increased to $300 \text{ kg N ha}^{-1} \text{ year}^{-1}$ (Table 2). Pasture eaten followed the same trend as pasture grown. Days in milk and milk production per cow did not change much between treatments explained by the decision to adjust comparative stocking rate to be approximately the same across treatments ($\sim 95 \text{ kg liveweight per t DM feed offered}$). Ground-truthing of the WFM showed that the model predicted milk production per cow matched the observed data reasonably well (Fig. 2). Milk production per hectare increased from N0 to N150 to N300 mainly because of the increase in stocking rate across the treatments (Table 2). Although predictions differed between the three models used, all showed increases in N leaching and GHG from N0 to N300 (Table 3).

The magnitude of annual N fluxes increased from N0 to N300, except for legume N fixation which decreased as N fertilizer input increased (Fig. 3). Because the data in Fig. 3 are the averages of three models and not all the pools and fluxes were available from the models, the numbers in the diagram do not sum in terms of inputs and outputs for the farm or the soil system. The N300 system appears to be the most “leaky” of the systems with N leaching being on average 72% and GHG emissions 67% higher than N0 (Table 3); while N output in product is only 54% higher than N0 (Fig. 3).

Carbon flows generally followed the same pattern as N because of the close link between these two nutrients. Pools and fluxes increased in magnitude as N fertilizer input increased, except in surface OM and fresh OM where pool sizes for N150 appeared to be smaller than for N0 (Fig. 4). This was because of lower average farm herbage mass in N150 at the end of the run-in year, which carried over into the measurement year with lower average herbage mass through winter into spring until pasture growth rates accelerated. From about October the average farm covers for both N0 and N150 systems tracked around 2500 kg

Table 2 DairyNZ Whole Farm Model predicted farm physical performance for the 2017–2018 farm season (1 June 2017–31 May 2018) for Canterbury dairy systems with different levels of N fertilizer input, 0, 150 or 300 kg N ha⁻¹ year⁻¹

N input	Stocking rate cows ha ⁻¹	Pasture grown kg DM ha ⁻¹	Pasture eaten kg DM ha ⁻¹	Imported silage kg DM ha ⁻¹	Days in milk	Milk production kg MS cow ⁻¹	Milk production kg MS ha ⁻¹
0	2.3	9600	7800	430	260	379	872
150	3.2	13,200	11,500	1634	262	370	1185
300	3.6	16,300	13,700	935	265	383	1378

MS milksolids (fat + protein)

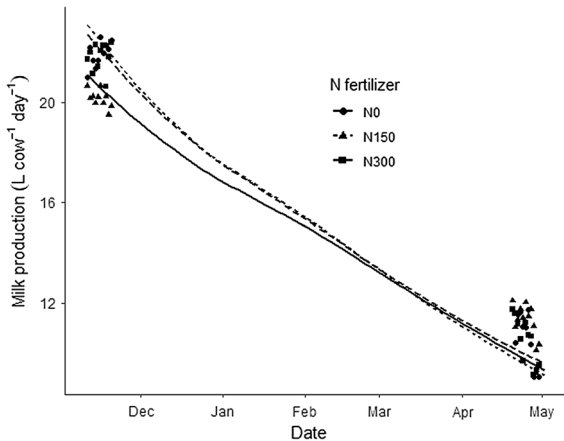


Fig. 2 Observed versus predicted milk production (L cow⁻¹ day⁻¹) for dairy systems with different levels of N fertilizer application, 0, 150 and 300 kg N ha⁻¹ year⁻¹. Symbols represent observed data for individual cows and lines represent herd averages as predicted by the DairyNZ Whole Farm Model

DM ha⁻¹. The lower average herbage mass in the first part of the season in the N150 system generated less senescence from above and below ground resulting in slightly lower average annual C pool sizes for fresh and surface OM (Fig. 4). The soil C pool is dominated by the size of the humus OM pool, with total C in the topsoil in the same ballpark (100–120 t ha⁻¹) as

measured by Parfitt et al. (2014) for non-allophanic dairy soils across New Zealand. Carbon input into the soil pool is dominated by litter from pasture flowing into the surface OM pool, with the largest respiration losses also from the surface OM pool (Fig. 4).

The lysimeter study

The leachates from the lysimeters (eight each from the N0, N150 and N300 paddocks, four with and four without urine addition) were analysed for nitrate and ammonium to estimate N loss from soils under the treated swards. The amount of mineral-N leached from the fertilizer only treatments did not differ, regardless of rate of fertilizer application (Table 4). The application of animal urine at cf. 700 kg N ha⁻¹ resulted in a significantly greater amount of mineral-N leaching loss compared with nil-urine treatments (Table 4). The application of fertilizer-N at 150 kg N ha⁻¹ in addition to urine led to a greater mineral-N leaching loss (157 kg N ha⁻¹) compared with the urine only treatment (76 kg N ha⁻¹) (Table 4). The application of fertilizer-N at 300 kg N ha⁻¹ in addition to urine did not result in a greater mineral N leaching loss (63 kg N ha⁻¹) compared with the urine only treatment (76 kg N ha⁻¹) (Table 4). The leaching losses

Table 3 Environmental metrics for Canterbury dairy systems with different levels of N fertilizer input, 0, 150 or 300 kg N ha⁻¹ year⁻¹. Nitrate (N) leaching and greenhouse

gas emissions are the predictions from three different models; the DairyNZ Whole Farm Model (WFM), DairyMod and Overseer®

N input	N leaching kg N ha ⁻¹ year ⁻¹			Greenhouse gas emissions t CO ₂ -eq ha ⁻¹ year ⁻¹		
	WFM	DairyMod	Overseer	WFM	DairyMod	Overseer
0	48	57	32	10.2	6.9	7.3
150	62	80	54	14.8	9.6	11.0
300	71	88	78	16.0	11.1	13.5

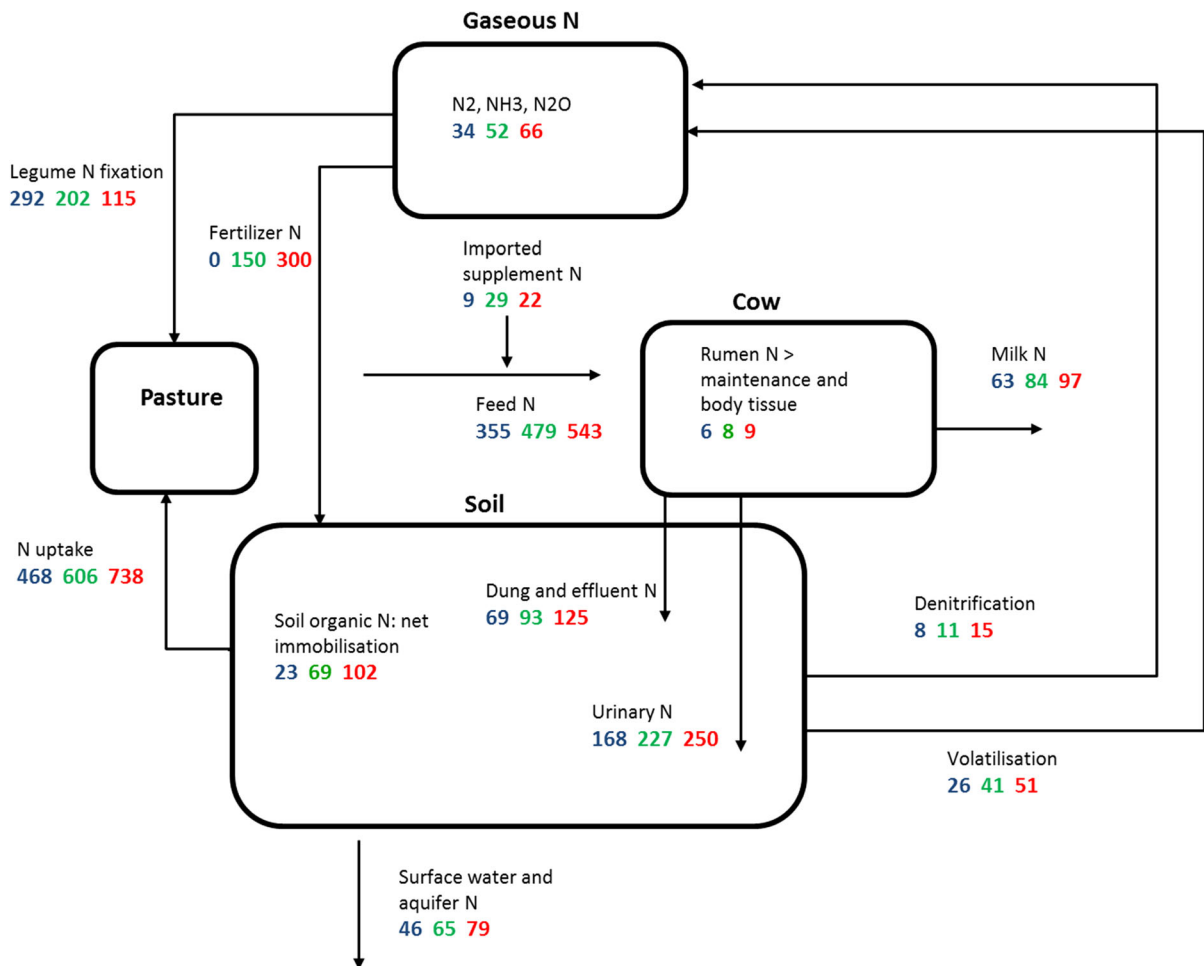


Fig. 3 Predicted nitrogen (N) fluxes ($\text{kg N ha}^{-1} \text{ year}^{-1}$) for Canterbury dairy systems in the 2017–2018 farm season with different levels of N fertilizer input; 0, 150 or 300 $\text{kg N ha}^{-1} \text{ year}^{-1}$. Numbers are from left to right N0 = blue; N150 = green; N300 = red. Soil represents the top 20 cm of a Lismore

soil. Results are averages of three models, the DairyNZ Whole Farm Model, DairyMod and Overseer[®]. Arrows are fluxes and boxes depict imaginary component boundaries. (Color figure online)

from the 300 kg ha^{-1} plus urine was lower than that from the 150 kg ha^{-1} plus urine treatment.

Discussion

Nitrogen surplus and use efficiency

The range of predicted values for N flows for the three systems in this study and modelled values for hypothetical pasture-based dairy systems were similar to values previously published (de Klein et al. 2010; Field and Ball 1981). We found that the N in animal

product, as a percentage of N eaten, was approximately 18%, which is comparable with the range reported for dairy cattle of 15–30% (Whitehead 1995). Nitrogen in urine ranged from 67 to 71% of total N excreted, consistent with the range of values reported in the literature for dairy cows eating feed of high N content and nutritive value (Jarvis et al. 1995). Inputs of N into farm systems (primarily from fertilizer, imported feed, and biological nitrogen fixation) always exceed outputs in animal products (mainly milk, but also sales of animals and feed); the difference is the whole-farm N surplus, some of which may be retained, but most of which is lost to the

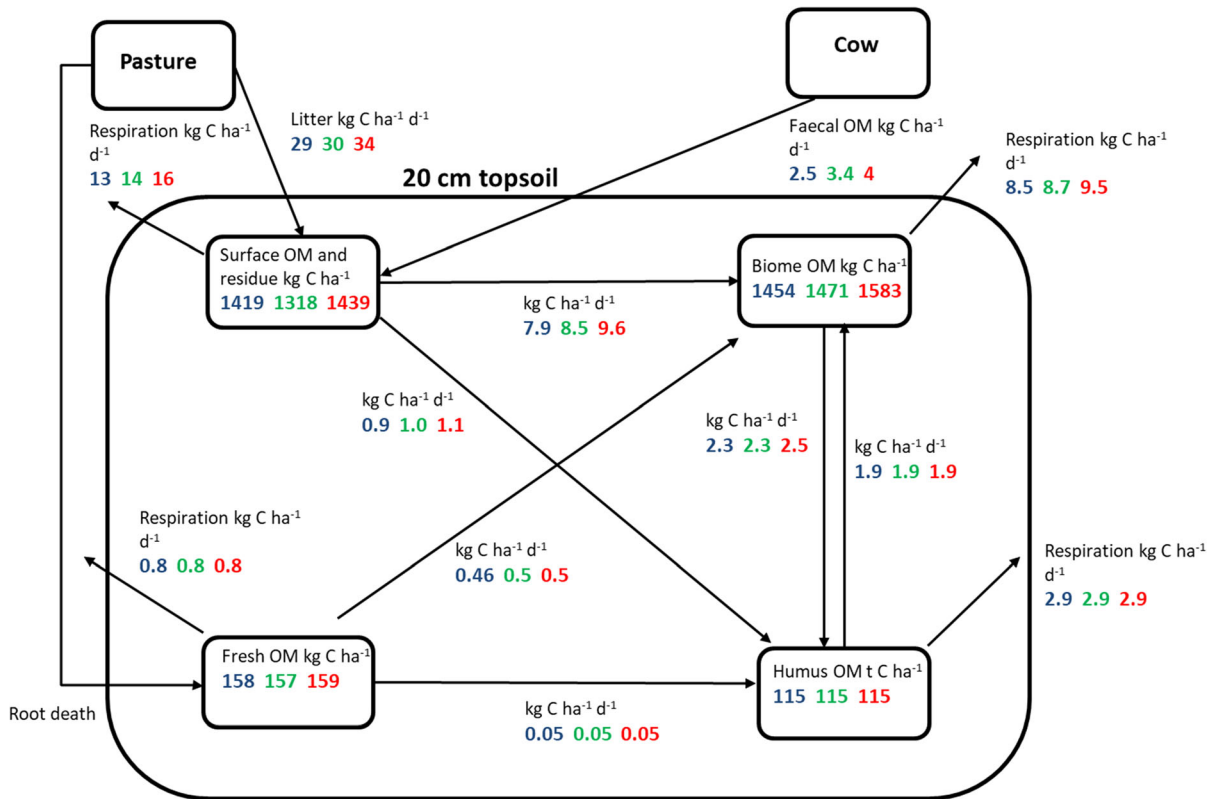


Fig. 4 Predicted carbon (C) pools (boxes inside the topsoil) and fluxes (arrows) in the top 20 cm of a Lismore soil for Canterbury dairy systems in the 2017–2018 farm season with different levels of N fertilizer input; 0, 150 or 300 kg N ha⁻¹ year⁻¹.

Numbers are from left to right N0 = blue; N150 = green; N300 = red. Results are from the DairyNZ Whole Farm Model and DairyMod. OM = organic matter. (Color figure online)

Table 4 Total amounts of N leaching recorded from lysimeters with three levels of fertilizer N input (0, 150 or 300 kg N ha⁻¹ year⁻¹) and either receiving a urination event of cf. 700 kg N ha⁻¹, or not

Treatments N fertilizer input; no urine/urine	Nitrate-N (kg N ha ⁻¹)	Ammonium-N (kg N ha ⁻¹)	Mineral-N (kg N ha ⁻¹)
0; no urine	0.2 ^a	0.2 ^a	0.4 ^a
0; urine	37.9 ^a	37.8 ^b	75.7 ^b
150; no urine	7.0 ^a	0.6 ^a	7.6 ^a
150; urine	125.8 ^b	31.6 ^{ab}	157.4 ^c
300; no urine	5.9 ^a	0.2 ^a	6.1 ^a
300; urine	59.0 ^{ab}	4.1 ^a	63.1 ^{ab}
LSD (<i>P</i> < 0.05)	69.51	36.53	68.16

Numbers followed by different superscript letters are statistically different in the same column (*P* < 0.05)

environment. Our N surplus (i.e. N inputs–N outputs) varied from 238, to 297, to 340 kg N ha⁻¹ for N0, N150 and N300, respectively. This can be compared

with the range of 40–700 kg N ha⁻¹ year⁻¹ reported for pastoral dairy farms in NZ (de Klein et al. 2017). Increasing the efficiency with which all N inputs are

converted to outputs (the ‘Nitrogen Use Efficiency’, or NUE) reduces the N surplus and the leaching risk. In our study the NUE was 21, 22 and 22% for N0, N150 and N300, respectively. These are at the low end of the NUE range reported for the Canterbury region, and partly explain the relatively large N surpluses, although these were comparable to those reported for Canterbury (Pinxterhuis and Edwards 2018). Since N surplus is positively correlated with N loss to the environment (Beukes et al. 2012), it was no surprise that the environmental N loss (leaching and gaseous) followed the same pattern as N surplus, lowest in the N0 and highest in the N300 treatment.

Nitrogen losses

The results from the lysimeter study supported the predicted increase in N leaching from N0 to N150, but not the increase from N150 to N300. It was not apparent what caused the lower mineral N leaching loss from the lysimeters treated with 300 fertilizer-N plus urine compared with the 150 fertilizer-N plus urine. The difference may simply be due to natural variability in soil physical conditions between these treatments. Nevertheless, the amounts of N loss from the fertilizer plus urine treatments are similar to the losses reported for similar treatments on free-draining soils (Di and Cameron 2002; Cameron et al. 2013).

A comprehensive systems experiment conducted in New Zealand comparing the effects of N inputs on N losses to the environment (Ledgard et al. 1999) (the ‘1.75 t MS ha⁻¹’ trial, which compared 0, 200 or 400 kg N fertilizer ha⁻¹ year⁻¹ in a farm study) showed that for a silt-loam soil the nitrate leaching increased by around 0.35 kg N ha⁻¹ for every 1 kg N ha⁻¹ increase in the annual N surplus. This relationship explained about 65% of the variation in nitrate leaching. In this case nitrate leaching was measured directly, not estimated with models. Using data from our study (Fig. 3) and regressing N leaching against N surplus for the three systems modelled, we found that N leaching increased by around 0.37 kg N ha⁻¹ for every 1 kg N ha⁻¹ increase in the annual surplus. Furthermore, our results for total N loss versus N output as food production confirm the pattern presented by Chapman and Parsons (2017), with annual rates of N loss to the environment increasing exponentially as the total food N production increases from 63 to 97 kg N ha⁻¹ year⁻¹ (from N0 to N300). From

N0 to N150 the increase in loss was 40% more than the increase in N product, while from N150 to N300 the increase in loss was 60% more than the increase in N product. This supports the results from Clark (1997) who compiled data from seven fertilizer-rate studies done on New Zealand dairy pastures from 1971 to 1995 where he showed that N use efficiency (kg N in milk per kg N fertilizer applied) was greatest at annual fertilizer application rates between 100 and 200 kg N ha⁻¹ year⁻¹, and declines steadily thereafter. The exponential shape of the N loss versus N product curve is the result of the production of the system becoming limited by light interception and C capture per ha per unit of N input (Parsons et al. 2016), by the capacity of the soil to store N becoming saturated, and by the fact that grass is harvested by animals that return a large proportion of the N ingested in the form of poorly distributed, highly concentrated, urine patches.

All three models predicted net immobilization of N in the soil organic matter pool (Fig. 3), which is typical for a moderately well-drained Recent Brown soil e.g. Lismore (Schipper et al. 2014). The magnitude of predicted N immobilization (69 kg N ha⁻¹ year⁻¹ for N150, Fig. 3) aligns well with the annual change in total soil N (0–10 cm) of 60 kg ha⁻¹ year⁻¹ measured for 85 dairy soils in New Zealand over a period of 7 years (Parfitt et al. 2014).

Carbon pools and flows

Our results suggest that the rate of C flux through the soil system is accelerated by adding more N fertilizer to the system, with very little change in the total C pool in the topsoil. Rates of litter and faecal C flows into the topsoil increased from N0 to N300, but so did rates of respiration losses from the surface OM and biome pools. This finding is supported by the literature showing that increasing rates of N fertilizer accelerates the C–N cycle in the soil, with greater uncoupling and potential losses from the system (Zhu et al. 2016). In a study of the corn-soybean agroecosystem in the Midwestern US, Russell et al. (2009) found that for maize, organic C inputs increased with N fertilization. Below ground net primary production (NPP), comprising only 6–22% of total maize NPP, was not significantly influenced by N fertilization. When all phases of the maize rotations were evaluated over the long term, organic C decay rates increased

concomitantly with C input rates. Increases in decay rates with N fertilization apparently offset gains in C inputs to the soil in such a way that soil C sequestration was virtually nil in 78% of the systems studied, despite up to 48 years of N additions. In a long-term fertilizer field trial in New Zealand, Wakelin et al. (2017) found that as the NPP increased with phosphorous addition, greater losses of C occurred concomitantly with increased phosphorous fertility, and microbial biomass increased with soil phosphorous status. Mineralization of C substrates added to soil also increased with soil phosphorous status. The results indicated potential conditioning of the microbial community for rapid C cycling.

Conclusion

Increasing N fertilizer inputs into pastoral dairy systems results in increasing pasture and animal productivity, but at a cost of greater inefficiencies, larger soil mineral N pools, and greater risk of N being lost from the system through leaching and greenhouse gas emissions. Therefore, striving for maximum N output in animal products as a goal is not a viable option from an environmental point of view. At the opposite end of the scale at low levels of N fertilizer input, surplus and losses are low, but so is N output in product. Striving for this goal means low production and profitability, which is also not a desirable outcome. There is no single optimum point on this continuum of N fertilizer input. Rather there is a zone where N output is relatively high, but not maximum, and N losses are relatively low. Setting the system up to operate within this zone is therefore a viable goal for a balanced outcome. For our study site this zone appears to be somewhere between 100 and 200 kg N ha⁻¹ year⁻¹. Understanding the relationship between N inputs and outputs for any individual pastoral dairy farm system is fundamental for setting the platform from which N fertilizer decisions can be managed effectively. We recommend that future research focuses on another potential control point for affecting N use efficiency in pastoral dairy systems: managing the N processing in the animal where N gets dissociated from organic matter, from where it has a much larger potential for environmental losses.

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