ORIGINAL ARTICLE

Nitrogen loss partitioning and emissions in intensive subtropical hybrid dairy systems

Sofía Stirling · **Fernando Alfredo Lattanzi** ·
Santiago Fariña · **Ronaldo Vibart**

Abstract Dairy systems in South America's humid subtropics include grass-legume pastures in rotation with winter-summer double-cropping for silage, thus combining direct grazing with periods of effective confinement for concentrate and silage supplementation. The environmental impacts of these so-called hybrid systems remain unclear. We compiled detailed nitrogen (N) circulation budgets for four dairy systems in Uruguay stocked at 1,300 kg liveweight ha^{-1} but with contrasting feeding strategies (lesser *vs.* greater use of maize silage) and cow genotypes (New Zealand *vs.* North American Holstein–Friesian) and then used a farm environmental model (Overseer® Science) to partition N surpluses into losses to water and air, and to estimate greenhouse gas (GHG) emissions. All systems exhibited substantial N surpluses (190–238 kg N ha^{-1}) and moderate whole-farm N use efficiencies (31–35%). Conversely, estimated GHG emission intensities were comparatively low: less than 9.9 kg CO_2e kg^{-1} milk fat+protein, with N_2O representing less than 20% of total emissions. Nitrogen surpluses were predicted to be lost mainly through leaching (43%) and volatilisation (41%), not denitrification (10%). Loafing pads and fallow periods in pasture-crop transitions contributed the most to N losses. Feeding strategy and cow genotype effects on these patterns were minor. This study identified (i) specific spatiotemporal spots with a disproportionally large impact on potential losses of N –for instance, 12–18% of the farm area accounted for 87–90% of predicted N leaching losses, and (ii) a limited influence of feeding strategy and cow genotype. Such insights into where critical environmental impacts reside provide a quantitative foundation for future studies on intensified hybrid subtropical dairy mitigation strategies.

Keywords Nitrogen losses · Leaching · Farm system · GHG · Simulation

S. Stirling (✉) · F. A. Lattanzi · S. Fariña
Instituto Nacional de Investigación Agropecuaria (INIA),
La Estanzuela. Ruta 50 Km 11, Colonia, Uruguay
e-mail: sstirling@inia.org.uy

R. Vibart
Grasslands Research Centre, AgResearch Limited, Private
Bag 11008, Palmerston North 4442, New Zealand

Introduction

Intensified pasture-based animal production systems worldwide face the challenge of reducing their environmental impact and enhancing standards on animal welfare, human health, and rural livelihoods and well-being without compromising their profitability (Britt et al. 2018; Clay et al. 2020). The projected growth in global demand for dairy products over the next 50 years (OECD/FAO 2021) requires the

scientific and innovation system to address this challenge (Rowarth and Parsons 2015).

Dairy systems from temperate humid regions, such as New Zealand, southern Australia and northern Europe, predominantly rely on grazing perennial ryegrass pastures with either minimal or time-limited supplementation, which helps their economic sustainability by maintaining low production costs (Oenema and Oenema 2021; Luo and Ledgard 2021; McDowell et al. 2022). However, intensified dairy farming via higher stocking rates and increased pasture utilisation has led to adverse environmental impacts, mainly due to water pollution through nitrogen (N) leaching from urine deposition on pasture, as well as increased greenhouse gas (GHG) emissions, primarily methane (CH_4) from enteric fermentation and nitrous oxide (N_2O) from N fertiliser and urine (Wachendorf et al. 2005; Roche et al. 2017; Rotz et al. 2020; Clay et al. 2020; Kelly et al. 2020).

Compared to temperate humid regions, dairy systems in humid subtropical areas experience higher radiation, temperature, and evaporative demand. These conditions provide a flexible window for double-cropping annual winter-summer C3-C4 crops, increasing primary productivity and water use, thus reducing percolation below root depth (Ojeda et al. 2018a, b). However, summer heat stress, often combined with short-term acute water deficits, becomes a severe restriction for the survival of perennial temperate grasses (Jauregui et al. 2017; 2024) and legumes (Scheneiter et al. 2019). Furthermore, heat waves, i.e., several days with sustained high humidity and temperature, impair grazing schedules, animal welfare and milk production (Morales-Piñeyrúa et al. 2022).

Specifically, dairy systems in Uruguay are characterised by sequences that combine sown pastures that last three to four years used mainly for grazing, followed by one or two years of double-cropping annual forage crops that are either grazed or, most frequently, conserved as silage (INALE 2019). Grazed pastures typically comprise a mixture of various temperate grasses and legumes, whereas maize and annual ryegrass or oats are usually the annual forage crops. Hence, a significant portion of the farm is out of the grazing platform, destined for silage production or undergoing fallow or pasture/crop establishment periods. Moreover, animal access to grazing is often restricted, to one daily grazing session (either morning or afternoon) or none, due to extreme weather events, such as heavy rains or heat waves. Therefore, grazed pasture rarely surpasses 60% of the annual dry matter intake (DMI), with the remaining of the diet consisting of silage and concentrates that animals consume off-paddock, typically in feed/loafing pads (Fariña and Chilibroste 2019; Lazzarini et al. 2019). McDowell et al. (2022) characterised these as "hybrid" systems that combine three to eight months of direct grazing with consumption of preserved forage during the non-grazing periods.

In Uruguay, clear opportunities have been highlighted for competitively lifting milk production of hybrid systems with five to nine net months of grazing through increases in stocking rate (Fariña and Chilibroste 2019; Lazzarini et al. 2019; Ortega et al. 2024) enabled by higher amounts of grazed pasture per hectare finely tuned with moderate supplementation rates (Gareli et al. 2023). Up to two-fold increases in stocking rates relative to the national average have been shown to improve economic performance and to be biologically feasible via different combinations of cow genotype (New Zealand vs. North American Holstein-Friesian) and feeding strategy (higher vs. lower proportion of silage in diets) (Stirling et al. 2021a, b). However, the extent to which such intensified systems impact the environment remains under scrutiny (Fariña and Chilibroste 2019; Darré et al. 2021).

Water pollution has become a concern in Uruguay, particularly regarding summer cyanobacterial blooms, which have been linked to changes in water temperature and pH (Beretta-Blanco and Carrasco-Letelier 2021), but also to nutrient concentration of waterways and agricultural land use (Chalar et al. 2017; Gorgoglione et al. 2020; Kruk et al. 2023). As a result, national policies that currently regulate land use seek to limit erosion, labile phosphorous concentration in the topsoil, and use of phytosanitary products (ROU 2008). Since N management plays a significant role in ensuring sustainable food production systems (Leip et al. 2021), it is critical to assess to what extent the pattern of environmental impacts observed in intensified temperate pasture-based dairy systems holds for rainfed hybrid systems in subtropical regions that have more intense cropping sequences, higher silage supplementation, and more variable rainfall regimes.

Leveraging knowledge from international experience can help identify the main environmental

risks that production systems pose in regions where empirical assessments are still scant. In this situation, adapting models extensively used in similar contexts appears valuable (Rowarth and Parsons 2015; Jones et al. 2017). In this study, we employ an integrative whole-farm simulation approach by using the Overseer® Science model, adapted from New Zealand's context, to provide an initial assessment of critical paths and sources of environmental impact of prospective intensified hybrid systems for humid subtropical areas in Mid-Atlantic South America.

The objectives of this study were, first, to compile detailed N circulation budgets for experimental hybrid dairy farmlets with contrasting feeding strategies (lower vs greater use of silage) and cow genotypes (New Zealand vs. North American Holstein-Friesian) that doubled milk production per hectare compared with Uruguay´s national average by increasing stocking rates, and second, to model both GHG emissions intensity and the relative partitioning of N surpluses into losses to water and air.

Materials and methods

The present study compiles detailed N budgets of four experimental dairy farmlets over two years. Then it uses the Overseer® Science Model (Science version 6.3.5; hereafter referred to as Overseer) (Wheeler et al. 2003), parameterised with local soil and climate data, to first evaluate how N surpluses are partitioned to water (nitrate leaching) and air (ammonia volatilisation, and N_2 and N_2O denitrification) and second, to estimate GHG emissions (CO_2, CH_4, and N_2O).

Farm systems experimental study

A detailed description of the farmlets is provided by Stirling et al. (2021a). Briefly, the farmlets were located at INIA La Estanzuela (34°20′S, 57°41′W, Uruguay) and aimed at increasing home-grown forage harvest to sustain a two-fold increase in stocking rate relative to the national average. Meteorological data was recorded at a station 1000 m from the experimental site.

The biophysical and economic performance was evaluated from June 2017 to May 2019 for the factorial combination of (i) two Holstein–Friesian cow genotypes, New Zealand (NZHF) and North American Holstein Friesian (NAHF), with mature body weights of 479 and 583 kg, respectively, and (ii) two feeding strategies that aimed to attain different proportions of grazed pasture vs. silage (pasture silage + maize silage) in the diet but with the same level of concentrate (5.3 Mg DM ha^{-1} year^{-1}). The Grass Fixed feeding strategy (GFix) diet had a fixed ratio of 1/3 pasture, 1/3 concentrate and 1/3 silage of total DMI (Table 1). All supplements were offered as a partial mixed ration on a feed pad. The Grass Maximum feeding strategy (GMax) was aimed at maximising direct grazing and thus had a flexible diet determined by pasture growth rate. In this strategy, silage was offered on a feed pad as a buffer in case of pasture shortage, and concentrate (1/3 of total DMI) was offered individually in the milking parlour.

Pasture-crop sequences comprised: (i) pastures, either pure tall fescue (*Festuca arundinacea*) or a mixed pasture with cocksfoot (*Dactylis glomerata*) and lucerne (*Medicago sativa*), (ii) annual winter fodder crops (oats *Avena sativa,* or annual ryegrass *Lolium multiflorum*), and (iii) maize (*Zea mays*) for whole-crop silage. In GFix, the pasture-to-crop ratio in the milking platform area (i.e., the area of the farmlet potentially grazable by the milking herd) was 60:40; in GMax, it was 80:20. Therefore, pastures lasted 3.5 years in GFix and 4 to 4.5 years in GMax. Stocking was always rotational, with daily strips grazed according to identical grazing management guidelines in all four treatments. Nitrogen fertilisation in grass-based pastures (tall fescue, annual ryegrass and oats) was applied after each grazing using urea (0.46 N) according to grazing rotation length (1 kg of N ha^{-1} per day of rotation, applied 48 h pre- or post-grazing; Rawnsley et al. 2014). Mixed pastures (cocksfoot with lucerne) were fertilised with N only from their third year onward, using the same rotation length rule. Maize crops were fertilised at sowing with 46 kg of N ha^{-1} and then re-fertilized at the V6 growth stage with 115 kg of N ha^{-1} using urea and organic fertiliser (*i.e.,* manure), following local recommendation for target yields of 14–15 Mg DM ha^{-1}.

Estimations made with overseer

The Overseer model was used to estimate annual nitrate leaching, denitrification, ammonia volatilisation, and GHG emissions of the four farmlets for two years. Overseer is a whole farm-scale model

Table 1 Summary of two years of biophysical results for four farmlets with either Grass Maximum (GMax) or Grass Fixed (GFix) feeding strategies and New Zealand (NZHF) or North American Holstein–Friesian (NAHF) cow genotypes (data from Stirling et al. 2021a)
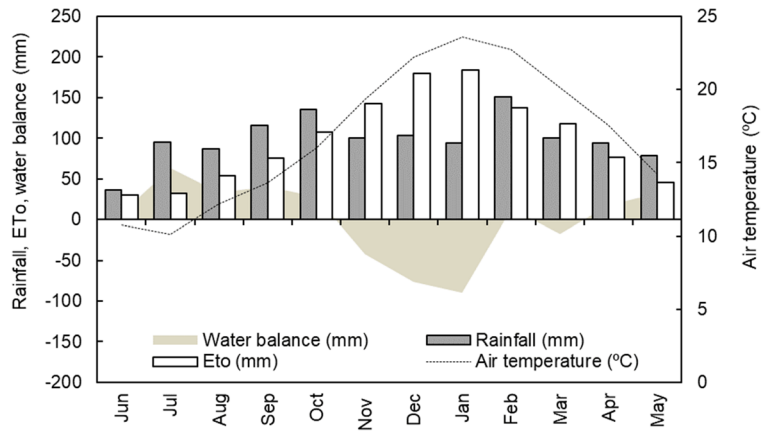
| | Farmlet | | | |
| | GMax | | GFix | |
	NZHF	NAHF	NZHF	NAHF
Farm				
Effective area (ha)	11.3	14.1	11.3	14.0
Number of cows	30	30	30	30
Stocking rate (cows ha^{-1})	2.7	2.1	2.7	2.2
Stocking rate (kg live weight ha^{-1})	1,402	1,229	1,413	1,266
Nitrogen fertilisation (kg ha^{-1} yr^{-1})	248	216	234	235
Production				
Milk solids (fat + protein) (kg ha^{-1} yr^{-1})	1,568	1,379	1,460	1,395
Milk fat (kg ha^{-1} yr^{-1})	862	771	786	773
Milk solids per cow (kg cow^{-1} yr^{-1})	521	616	543	602
Feed consumption (Mg DM ha^{-1})				
Pasture	6.3	5.2	4.2	3.7
Pasture silage	1.0	1.0	1.7	1.7
Maize silage	3.1	3.0	4.2	4.1
Bought-in concentrate	5.3	4.7	5.6	5.5
Total feed consumed	15.7	13.8	15.7	15.0

that develops budgets for major soil nutrients (N, P, K, S, Ca, Mg and Na) and calculates nutrient losses and on-farm GHG emissions (CO_2, CH_4 and N_2O). The model is constructed from empirical sub-models, with coefficients derived from data from New Zealand field experiments. The model has already been used to assess production systems in Uruguay (Becoña López et al. 2013) and Argentina (Bilotto et al. 2019). Nitrogen inputs include fertilisers, supplements, rainfall, and biological fixation. Nitrogen inputs from brought-in supplements include N in supplements brought to the farm or stored, N in animal health supplements, or net imports of N into a house block (to balance those removed as sewage loading). The amount of biological fixation in legumes (clover and lucerne), in the absence of added N inputs, is calculated from annual pasture production, average legume content in the pasture, and an N fixation rate. Nitrogen outputs include sold products (milk or meat), N transferred as animals, and effluents exported. The model calculates the N surplus—at the farm (farmlet) scale—by subtracting total N output from total N input and NUE as the ratio of total N output to total N input. In the present study, we refer to (1) *whole-farm* NUE, defined as total N exported from the farm and expressed as a percentage of total

N inputs from fertilisers, supplements, rainfall, and biological fixation, and (2) *animal* NUE, defined as N intake converted into animal products, and expressed as a percentage of N intake by the animal from crop, pasture and imported feed (de Klein et al. 2017).

The model was parameterised using local climate and soil data. Furthermore, most of the N fluxes were provided to the model from measured farm biophysical characteristics, namely animal size and number and their spatial and temporal distribution in pastures, milking parlour and feed pads, feed input from grazed pasture and silage supplement and concentrate, pasture and crop management, fertilisation rate and timing, and effluent and manure management. Rainfall, reference Penman–Monteith evapotranspiration (ETo), and temperature were entered as monthly inputs using 10-yr averages from the local meteorological station (Fig. 1). Soil texture, soil plant available water holding capacity, and topsoil and subsoil chemical and physical parameters were defined using data from a detailed soil map of the experimental site. The soil type, corresponding to a Vertisol in the USDA Taxonomy, was specified as melanic sedimentary with a silt-clayey loam texture for the topsoil and clayey texture for the subsoil, with 74 mm of plant available water holding capacity in the top 0.6 m

(effective rooting zone; see Supplementary material, Fig. S1). According to the model ranges, the slope range selected was from 0–7%. Average soil analysis values for Olsen P, MAF Quick Test (QT) K, Ca, Mg, Na, and S were entered for each farmlet. Generally, pastures and crops were not limited by P, K or S supply but experienced some degree of N deficiency. The milking platform comprised the areas of pasture and annual fodder crops, referred to in Overseer as the "pasture block" and the "annual crop block", respectively (Fig. 2). The latter includes the corresponding fallow periods before crop establishment and after

the crops were harvested. For the GMax feeding strategy, the "pasture block" represented 75% of the total area and the "annual crop block" 19%. In contrast, for the GFix feeding strategy, the pasture block represented 57% of the total area and the annual crop block 38%. The sum of laneways, loafing pad, milking shed and effluent storage systems, referred to as "off-grazing block" in this paper, was 6% of the area in all farmlets.

The dairy effluent system modelled included a holding pond, with prior separation of solids from liquids, and was exported off-farm. Following de
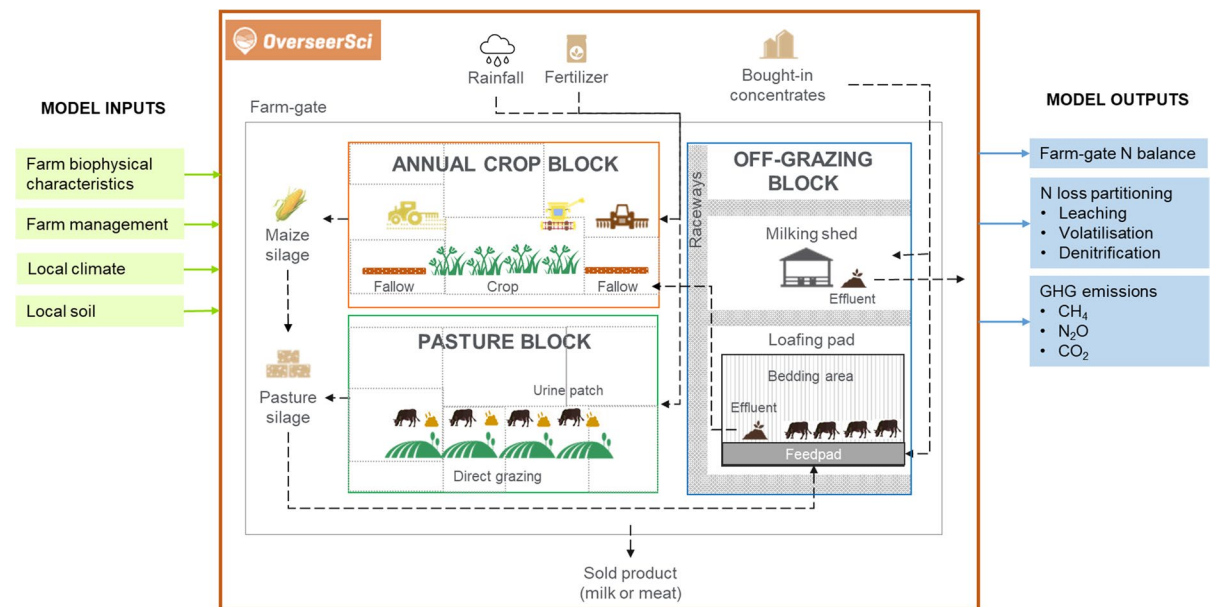
Fig. 2 Schematic depiction of the inputs, processes, and model outputs of the Overseer® Science model

Klein et al. (2017), this was considered a "negative input" rather than an output. The uncovered loafing pad is the area where animals were withheld from pasture for extended periods, and supplementary feeds were offered. As the cows spent a high proportion of the time on the loafing pad, the area presented (i) a bedding area for the cows to lie down and (ii) a feeding area with a concrete feed pad and floor where supplementary feed was offered. The bedding area presented a ground surface (inert surface) without effluent capture or management. On the feed pad area, the manure was scraped and stored in a stack with no water. This manure remained in a stack uncovered and was applied twice a year as fertiliser during the spring-fallow period (September and October) before the maize was sown. The manure was sampled for dry matter (45% DM) and N (3%) content. On average, 16 kg N ha^{-1} per year were applied as manure, representing around 7% of annual N applied per ha. Outputs from Overseer reported in this paper include the partition of N surplus into losses as nitrate leaching, ammonia volatilisation and denitrification, and on-farm annual CH_4, N_2O and CO_2 emissions, expressed in CO_2 equivalents per unit land area, per cow, and per unit of product.

Nitrogen losses

Nitrogen losses to water (leaching, runoff, direct discharge) and the atmosphere (volatilisation and denitrification) were modelled independently for the pasture block, the annual crop block, and the off-grazing block of each farmlet.

Nitrate leaching below 0.6 m (*i.e.,* "below the effective root zone") is modelled using transfer functions based on monthly drainage and nitrate concentrations in the soil solution. Drainage, which depends on the hydric balance and thus on evapotranspiration, precipitation and soil water holding capacity, was an input (Fig. S2), but the concentration of inorganic N in the soil solution, which depends on the balance between immobilisation, volatilisation, denitrification and pasture N uptake, was modelled (Selbie et al. 2013). Leaching is modelled separately for urine patches and other sources (such as faeces and applied fertiliser, effluents, and other organic additives), which are collectively referred to as "background". For urine, Overseer calculates the monthly deposition of excreta on grazeable areas and calculates the

amount leached using a transfer coefficient based on the pore volume of drainage. For background leaching, the model assumes that the pasture efficiency retaining N applied as fertiliser or effluent is generally high but declines in winter.

The proportion of N lost via ammonia volatilisation from urine patches varies between 0.10 and 0.20 depending on monthly air temperature and wind speed, evapotranspiration and rainfall, and soil temperature, moisture, and pH (adjusted by the effect of urea hydrolysis, Selbie et al. 2013). Ammonia volatilisation from fertiliser applications includes the effects of daily precipitation for the month, average monthly temperature, average monthly soil moisture to 0.6 m (relative to field capacity), a crop cover factor, soil sand content, and the type and rate of fertiliser applied.

Denitrification losses are estimated monthly using an emission factor adjusted for pugging and depending on soil type, water-filled pore space and temperature for each day of the month.

To assess the impact of climate interannual variability on the partitioning of N losses, each feeding strategy was modelled for 16 meteorologically contrasting years taken from historical weather data at the experimental site. Hence, rainfall, ETo, temperature and drainage were entered as monthly inputs in 16 independent model runs. Then, a regression analysis was performed to assess the sensitivity of the amount of N lost in each pathway and block. Finally, fitted regression equations were used to estimate the frequency distribution of N losses using the entire meteorological database (1967–2021).

GHG emissions

Overseer estimates the farm's CH_4, N_2O and CO_2 emissions per source and unit product, expressed in kg of CO_2 equivalent (CO_2e) per hectare per year, until the product is ready to leave the farm for processing, thus including embodied emissions of feeds and fertiliser brought into the farm and the emissions of the activities undertaken on the farm. The GHG model built within Overseer (Wheeler et al. 2008) is based on algorithms that include IPCC (2006) emission factors as used for New Zealand's Agricultural National Inventory (MFE 2018) and modified to include on-farm management practices (Wheeler et al. 2003).

79

The sources of CH_4 emissions are separated into enteric, dung and effluent. Enteric CH_4 emissions are calculated by multiplying an enteric CH_4 emission factor (MFE 2018)×total DMI (estimated monthly from the animal's metabolisable energy (ME) requirement and feed ME content, measured weekly in each farmlet). Dung CH_4 emissions are based on dung production estimated monthly as the sum of DMI × (1—digestibility) of each dietary component, accounting for the number of animals and diet quality. Methane emissions from stored effluent vary depending on the effluent management system, including storage and field application methods.

Nitrous oxide emissions are based on emission factors for direct N_2O losses from excreta in the paddock and effluent excreta, N fertiliser, crop residues, and indirect emissions from leached N or volatilised ammonia from the farm (MFE 2018).

The CO_2 emissions include electricity and fuel, and the indirect contribution of supplements and fertilisers processing and manufacturing accounted for using a "cradle to the farm gate" partial life cycle assessment.

Statistical analysis

Statistical analysis was carried out using JAMOVI software (The JAMOVI project 2023, Version 2.3). The farm-gate N balance variable and animal N intake were analysed using mixed models.

$$Y_{ijk} = \mu + YR_i + FS_j + G_k + (FS \times G)_{jk} + \varepsilon_{ijk}$$

Where Y_{ijk} is the dependent variable, μ the overall mean, YR_i is the random effect of the year (i = 2 yr), FS_j is the fixed effect of feeding strategy (j = GFix or GMax), G_k is the fixed effect of genotype (k = NZHF or NAHF), $(FS \times G)_{jk}$ is the fixed effect of the interaction between feeding strategy and genotype effects, and ε_{ijk} is the residual error (random). Fixed effects were declared statistically significant at P ≤ 0.05 and discussed as trends for 0.05 < P ≤ 0.10.

Results

Farm-gate nitrogen balance

Total annual farm N inputs (net of exported effluents from the milking shed) averaged 315 kg N ha^{-1} across farmlets and years (Table 2, Fig. 3). There was a trend for systems with NZHF cows to increase total N input under the GMax feeding strategy (interaction FS * G; $P = 0.053$). However, there was no effect of feeding strategy or genotype, nor their interaction on the different farm N inputs, N outputs, N surplus, and NUE.

In all cases, N from fertiliser and bought-in concentrates were the primary inputs, accounting for 51% and 47% of total N inputs, respectively. Biological N fixation and atmospheric deposition were minor fluxes, always less than 2% of total N inputs. Total N output in milk and livestock averaged 103 kg N ha^{-1} per year, without difference between systems.

Net farm N surplus averaged 212 kg N ha^{-1} year^{-1}, and whole-farm NUE averaged 33%. The ratio of N export to N intake, a measure of the intrinsic *animal* NUE, was much lower than the *whole-farm* NUE (24 *vs.* 33%, respectively, averaged across farms). In contrast, the ratio of N intake to N input, an index of N cycling intensity, ranged from 1.38 to 1.46. Both ratios indicate substantial recycling of N within all systems. No apparent differences were observed between cow genotypes or feeding strategies regarding the N budget results (Table 2).

Composition and distribution of N fluxes within farmlets

No differences between systems were observed in N intake from imported supplements and pasture silage (Table 2, Fig. 3). However, feeding strategy affected N intake from maize silage and direct grazing. In the GMax feeding strategy, cows consumed more N from pasture (208 *vs.* 142 kg N ha^{-1}, respectively; $P < 0.003$) and less N from maize silage (41 *vs.* 56 kg N ha^{-1}, respectively; $P < 0.005$) than in GFix. There was also an effect of cow genotype: NZHF cows consumed more N from pasture than NAHF cows (189 *vs.* 161 kg N ha^{-1}, respectively; $P = 0.043$).

In the GMax feeding strategy, 49% of the total N intake was attributed to pasture, with maize silage

Table 2 Whole-farm nitrogen (N) fluxes measured, and N losses predicted by the Overseer® Science model parameterised with local soil and climate data for four farmlets with contrasting feeding strategies (FS) [maximum consumption from grazed grass (GMax) *vs.* fixed contribution of grazed grass to 33% of the animal diet (GFix)] and cow genotypes (G) [New Zealand (NZHF) *vs.* North American Holstein–Friesian (NAHF)]

| | Farmlet | | | | | Significance[1] | | |
| | GMax | | GFix | | | | | |
Item	NZHF	NAHF	NZHF	NAHF	SEM	FS	G	FS x G
Farm N budget (kg N ha^{-1} yr^{-1})								
N INPUTS	343^x	312^y	292^y	315^y	11.1	**0.053**	0.625	**0.052**
Fertiliser	201	190	170	200	13.3	0.429	0.450	0.159
Supplements	179	168	178	175	8.0	0.536	0.202	0.475
Rainfall	2	2	2	2	0.0	1.000	1.000	1.000
Biological fixation	3	5	3	1	0.8	0.314	0.877	0.213
Effluent exported	-60	-53	-62	-63	4.3	0.347	0.675	0.512
N OUTPUTS (as product)	106	102	102	104	1.5	0.759	0.759	0.380
N BALANCE	238	210	190	211	12.1	0.112	0.776	0.103
Whole-farm NUE (%)	31	33	35	33	1.38	0.278	0.893	0.379
Cycling intensity (Nintake: Ninput)	1.39	1.35	1.46	1.29	0.035	0.793	0.051	0.159
N fluxes (kg N ha^{-1} yr^{-1})								
N intake from pasture	229	187	150	135	4.9	**0.003**	**0.043**	0.269
N intake from pasture silage	24	23	41	40	6.9	0.155	0.900	0.980
N intake from maize silage	43	40	57	56	1.8	**0.006**	0.411	0.827
N excreta[2]	368	317	325	301	6.7	**0.022**	**0.011**	0.142
Animal NUE (Nexport/Nintake, %)	22	24	24	26	0.8	0.206	0.102	0.778

[1]Values in the same row with different superscripts tend to differ (x–y; $0.05 < P < 0.10$)

[2]N excreta: N intake – N output (as product)

accounting for 7% and pasture silage 2% of the total N intake, whereas, in the GFix feeding strategy, N intake from pasture accounted for 35% of the total N intake, followed by maize silage at 12% and pasture silage at 7%.

Feeding strategies also affected the spatial localisation of N fertiliser inputs. In the GMax feeding strategy, 77% of the total imported N fertiliser was applied to the pasture block, while the remaining 23% was applied to the annual crop block. In contrast, within the GFix feeding strategy, 58% of the total N fertiliser was applied to the pasture block, with the remaining 42% allocated to the annual crop block (Fig. 3). Conversely, both feeding strategies had the same (comparatively small) amount of manure applied in the annual crop block.

Partitioning of N losses: critical pathways, farm areas and times of the year

Total predicted N loss averaged 247 kg N ha^{-1} per year across treatments (Table 2; Fig. 3; Fig. 4). The off-grazing block –comprising raceways, loafing pad, milking shed and effluent storage systems– was the primary source of N lost, accounting for 63% and 58% of total losses in GMax and GFix feeding strategies, respectively, attributed almost equally to leaching and volatilisation in both feeding strategies (48 and 43% in GMax, and 41 and 45% in GFix, respectively).
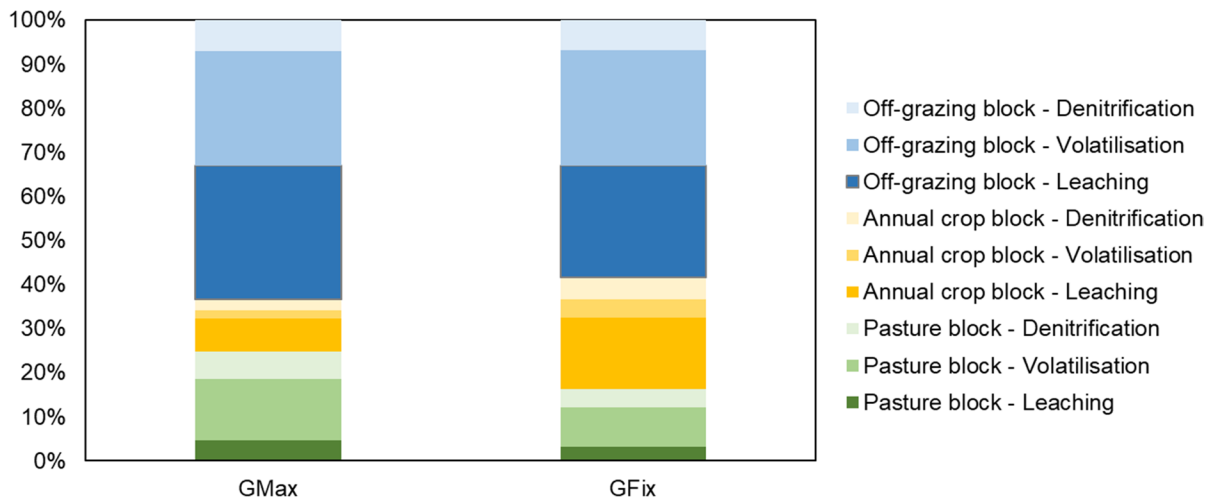
The pasture and annual crop block areas contributed differently to total N losses in the GFix and
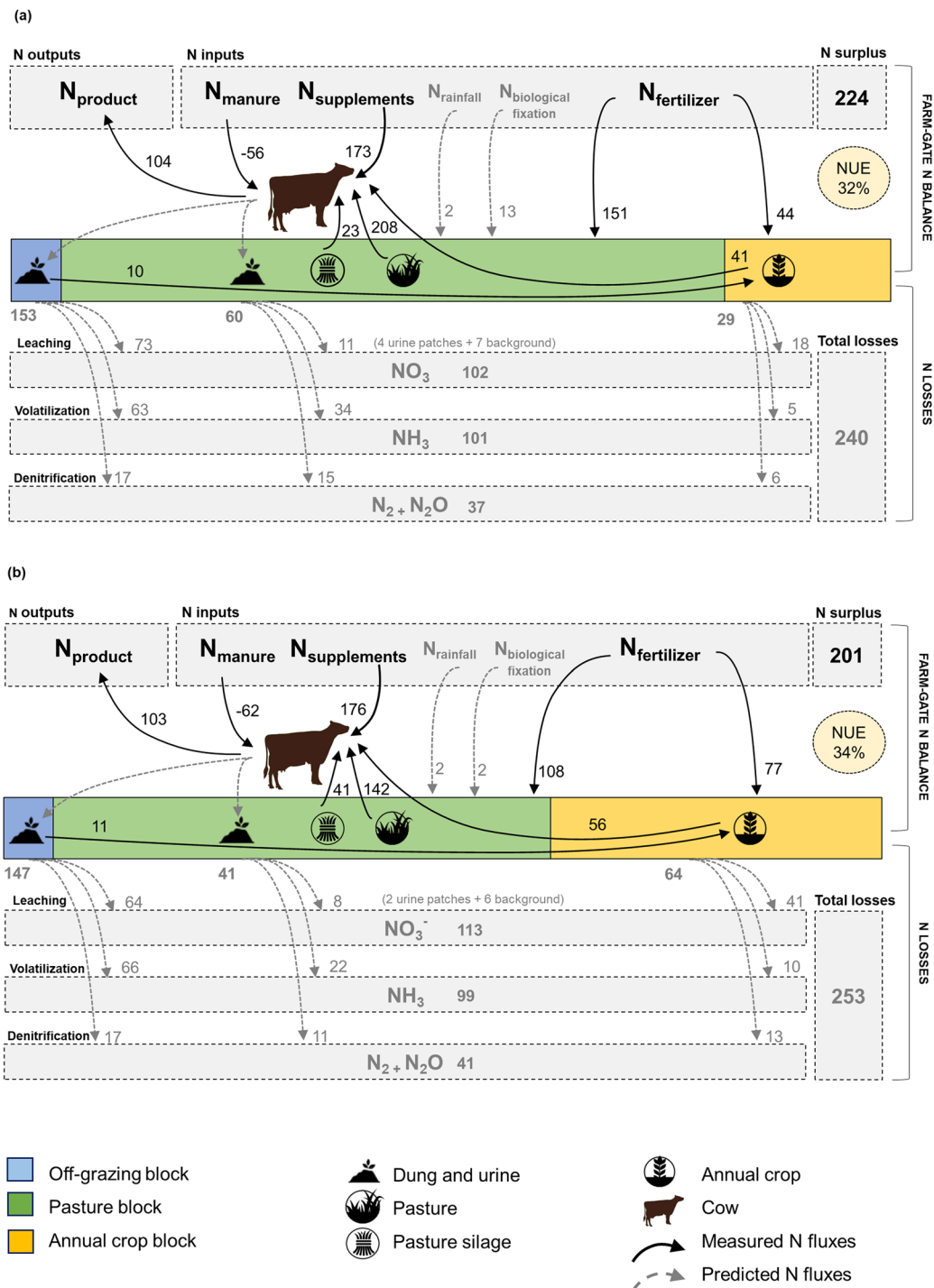
Fig. 3 Nitrogen (N) measured fluxes (continuous lines, black font) and predicted losses by Overseer® Science model (dotted lines, grey font) for farmlets with contrasting feeding strategies (a) maximum consumption from grazed grass (GMax) *vs.* (b) fixed contribution of grazed grass to 33% of the animal diet (GFix). The area of the pasture block is depicted in green, annual crop block in orange, and off-grazing block in blue. All fluxes are expressed in kg of N per hectare of farmlet (not block) per year. Total and subtotals are depicted in bold font

Fig. 4 Proportional contribution of different N losses (leaching, volatilisation and denitrification) predicted by the Overseer® Science model in the pasture block (green), annual crop block (orange) and off-grazing block (blue) for farmlets with contrasting feeding strategies: maximum consumption from grazed grass (GMax) *vs.* fixed contribution of grazed grass to 33% of the animal diet (GFix)

GMax feeding strategies because of the contrasting proportions. In GFix, the annual crop block was the second source of total N losses, accounting for 25% of total N losses, whereas GMax accounted for 12% of total N losses. Conversely, in GMax, the pasture block was the second source of N losses, contributing 25% of the total N losses compared to 16% in the GFix feeding strategy.

Within the annual crop block, nitrate leaching was the major N loss pathway (63–64% of total losses), whereas in the pasture block, volatilisation was the main N loss pathway (49–57% of total losses). For the annual crop block, nitrate leaching losses occurred during fallow periods before planting the maize (Sept-Oct) or the winter grasses (Feb-Mar) (Fig. 5a). Instead, in the pasture block, nitrate leaching from background losses was evenly distributed between winter, early spring, and autumn (Fig. 5b).

The (time-weighed) proportional contribution to total N leaching of the different farm areas was highly unequal, approaching the Pareto rule: 87 to 90% of losses were explained by 12 and 18% of the total (time-weighed) farm area in GMax and GFix, respectively, corresponding to the off-grazing block plus the fallow period of the annual crop block (Fig. 6). Within the pasture block area, urine patches occupy 30 and 18% of the total time-weighed area, explaining 6 and 5% of total N leaching losses for GMax and

GFix, respectively. Pasture and cultivated annual crop background losses occurred over 58% and 64% of the area but explained just 7 and 6% of total losses in the GMax and GFix feeding strategy.

Greenhouse gas emissions

On average, the total farm GHG emission was 14.5 Mg CO_2e ha^{-1} yr^{-1}, ranging from 13.8 to 15.3 Mg CO_2e ha^{-1} yr^{-1} (Table 3). The primary GHG was CH_4 (56–58% of the total GHG emitted), followed by CO_2 (22–25% of the total GHG emitted) and N_2O (19–20% of the total GHG emitted) (Fig. 7).

Total annual GHG emissions from farmlets with NZHF cows were higher than from farmlets with NAHF cows (15,067 *vs.* 13,978 kg CO_2e ha^{-1} yr^{-1}, respectively). Conversely, GHG emissions per cow were higher for the NAHF cows than the NZHF cows (~ 7 *vs.* 6 Mg CO_2e cow^{-1}). However, emissions were similar when expressed per kg of milk solids (9.8 kg CO_2e kg^{-1} milk fat + milk protein).

The primary source of CH_4 emission was enteric fermentation (Fig. 7), followed by CO_2 emissions from the production, transport, and processing of bought-in supplements. These two sources explained almost three-quarters of total GHG emissions. For N_2O, the primary source was the excreta deposited in paddocks and effluent (44–51% of the total N_2O
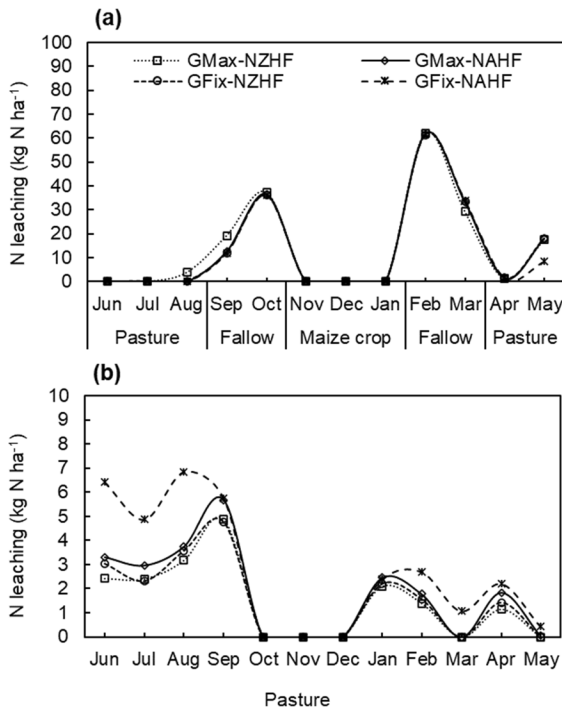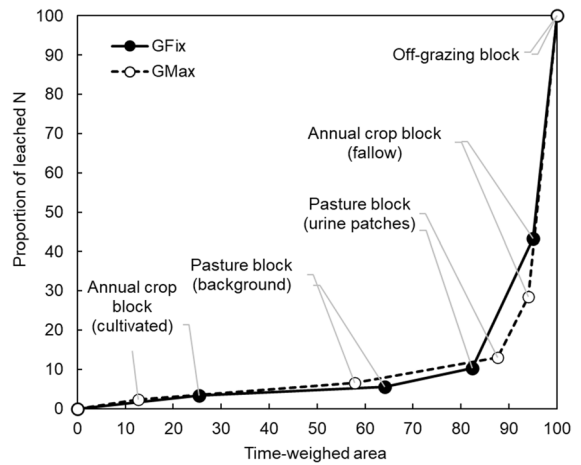
Fig. 5 Monthly background nitrate leaching (i.e., not from urine patches, kg N ha^{-1}) predicted by the Overseer® Science model in the annual crop block (a) and in the pasture block (b), in four farmlets with contrasting feeding strategies [maximum consumption from grazed grass (GMax) *vs.* fixed contribution of grazed grass to 33% of the animal diet (GFix)] and cow genotypes [New Zealand (NZHF) *vs.* North American Holstein–Friesian (NAHF)]

Fig. 6 Proportion of leached nitrogen (N) relative to the time-weight area from the different areas of the farm (pasture block, annual crop block and off-grazing block) predicted by Overseer® Science model in farmlets with contrasting feeding strategies [maximum consumption from grazed grass (GMax) *vs.* fixed contribution of grazed grass to 33% of the animal diet (GFix)]

emitted), followed by indirect emissions (29–35%) and emissions from synthetic N fertiliser application (17–21%).

Discussion

A comprehensive whole-farm approach was used to assess the relative magnitude of various potential environmental impacts of four intensification paths, all based on increasing the carrying capacity (i.e. stocking rate) via enhanced home-grown forage. Such a strategy is envisaged to maintain or increase the global competitiveness of rainfed dairy systems in humid subtropical Uruguay (Fariña and Chilibroste 2019). The approach combined (i) detailed measurements of all N fluxes within farmlets, except for (ii) excreta distribution and N losses estimated with model-based partitioning of the measured N surplus into spatial-, temporal- and biochemical-explicit losses, and (iii) model-predicted GHG emissions from CH_4, N_2O, and CO_2.

No specific regional modelling tools are yet available for environmental assessments in Uruguay. Therefore, we adapted the Overseer model from New Zealand's farm systems, parameterised it with local soil and climate data, and constrained it by providing all biophysical fluxes except losses and excreta partitioning. The limited validation of Overseer in subtropical climates warrants caution regarding predicted values, however, inferences in this study serve as an initial identification of prominent areas, time periods, and biochemical paths of N losses and as a quantitative profile of GHG sources. This should help weigh the relative magnitude of the various potential environmental impacts of intensified hybrid systems in Uruguay. Using experience from elsewhere when local data is insufficient is not uncommon in whole-farm analyses (*e.g.* Hoekstra et al. 2020).

Nitrogen loss partitioning in intensified subtropical dairy systems

Irrespective of cow genotype or feeding strategy, a significant amount of the N input was not converted

Table 3 Annual farm-scale GHG emissions by source predicted by the Overseer® Science model parameterised with local soil and climate data for four farmlets with contrasting feeding strategies [maximum consumption from grazed grass (GMax) *vs.* fixed contribution of grazed grass to 33% of the animal diet (GFix)] and cow genotypes [New Zealand (NZHF) *vs.* North American Holstein–Friesian (NAHF)]

| | Farmlet | | | |
| | GMax | | GFix | |
	NZHF	NAHF	NZHF	NAHF
GHG emissions per unit area (kg CO_2e ha^{-1} yr^{-1})	15,367	13,850	14,766	14,105
Methane (CH_4)	8,789	7,974	8,435	7,899
Enteric	8,689	7,882	8,327	7,806
Dung	66	63	48	45
Effluent	34	29	60	49
Carbon dioxide (CO_2)	3,439	3,231	3,537	3,515
Supplements (bought-in concentrate)	2,282	2,121	2,368	2,276
Nitrogen fertiliser (inorganic)	770	718	636	739
Fertiliser and organic inputs	119	128	141	127
Fuel	88	81	128	121
Electricity	58	47	58	47
Other	123	138	206	206
Nitrous oxide (N_2O)	3,140	2,645	2,795	2,691
Excreta (deposited in paddocks and effluent)	1,609	1,335	1,324	1,189
Indirect emissions	963	769	982	940
Nitrogen fertiliser (inorganic)	567	538	483	556
Crops	2	3	7	7
GHG emissions per animal				
kg CO_2e cow^{-1}	6,268	6,943	6,026	7,068
GHG emissions per unit product				
kg CO_2e kg fat protein corrected milk^{-1}	0.77	0.76	0.76	0.75
kg CO_2e kg milk solids^{-1}	9.6	9.7	9.8	9.9
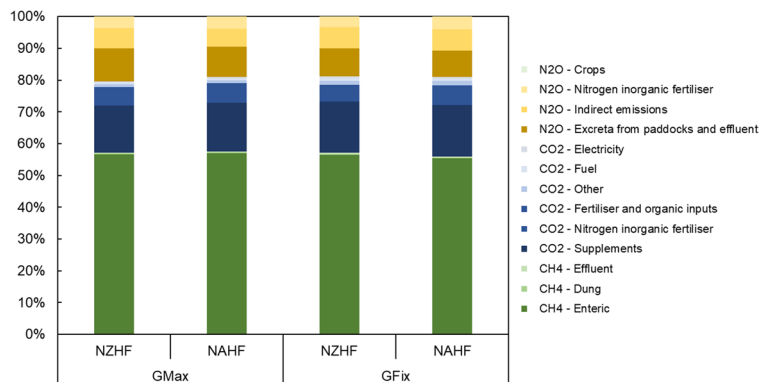
Fig. 7 Proportional contribution of different gases and sources to farm-scale GHG emissions predicted by Overseer for four farmlets with contrasting feeding strategies [maximum consumption from grazed grass (GMax) *vs.* fixed contribution of grazed grass to 33% of the animal diet (GFix)] and cow genotypes [New Zealand (NZHF) *vs.* North American Holstein–Friesian (NAHF)]

into milk and was, therefore, liable to be lost to the environment. The average N surplus across the four intensification strategies was 212 kg N ha^{-1} yr^{-1}, comparable to values reported for intensified pasture-based dairy systems from temperate regions such as New Zealand (Luo and Ledgard 2021; 186–281 kg N ha^{-1}), Australia (Gourley et al. 2012; 193 kg N ha^{-1}), Ireland (Buckley et al. 2016; 155 kg N ha^{-1}), and the Netherlands (Oenema and Oenema 2021; 174–208 kg N ha^{-1}). These results confirm a trend well-established in the literature that increased stocking rates in pasture-based systems lead to improved milk production and economic profit and result in higher farm-gate N surpluses (Powell et al. 2010; de Klein et al. 2017). Therefore, environmental challenges associated with intensified pasture-based dairy systems in humid subtropical climates might be as pronounced regarding N surplus as those observed in temperate regions, namely higher losses of reactive N species to waterways and the atmosphere.

Disaggregating N losses into the different sources, including urine and dung excretion in the field, fertiliser application and manure storage, has been the focus of recent research (*e.g.* Fischer et al. 2016; Hyde et al. 2016; Harty et al. 2016; Roche et al. 2016; Forrestal et al. 2017). However, fewer studies have evaluated N loss pathways at a whole-farm level. Indeed, to our knowledge, this study is the first approximation to a quantitative whole-farm partitioning analysis of the particular chemical forms, areas within a farm, and times through the year into which N surplus would be lost in dairy systems in subtropical humid climates. Further, we compared the effect of feeding strategy and cow genotype, on which even examples from temperate regions are scant, and thus, opportunities for comparison are scarce.

In all evaluated farm systems, leaching and volatilisation were the major N loss pathways (44% and 41% of the total N losses, respectively), while denitrification contributed a smaller portion. This contrasts with pasture-based dairy systems located in temperate regions such as Ireland (Burchill et al. 2016; Hoekstra et al. 2020) or New Zealand (Luo and Ledgard 2021), where ammonia volatilisation has been identified as the most significant N loss pathway followed by nitrate leaching from urine patches. The higher ammonia volatilisation levels between systems and regions could be explained by the presence of off-paddock (Luo and Ledgard 2021)

or housing (Hoekstra et al. 2020) facilities with adequate infrastructure to collect the effluent produced. This, in turn, leads to greater storage and utilisation of slurry as fertiliser, which is a clear avenue for the improvement of Uruguayan systems. Another reason is that in this subtropical region, the climate seems to exert a major effect on leaching but not on gaseous losses. The sensitivity analysis shows leaching is predicted to vary significantly in response to water balance (Fig. S3). Had our study been carried out in years with drier winters and autumns, volatilisation would have become the main N loss path. These interactions between sites and climate reinforce the value of whole-farm modelling to compare systems quantitatively.

Loafing pads and fallows - major spots of potential N leaching

Around 60% of the total N losses in the studied systems occurred from the ~6% farm area occupied by feed/loafing pads, raceways, and effluent stores (i.e. the off-grazing block). This contrasts with results from temperate pasture-based dairy systems in which urinary N deposited onto pastures is commonly identified as the primary source of N leaching (*e.g.* Clark et al. 2007; de Klein et al. 2010). Notably, in New Zealand pasture-based dairy systems, only 5 to 10% of the excreta is deposited on concrete in the milking sheds and neighbouring yards (Luo and Ledgard 2021). Furthermore, leaching within off-paddock facilities is minimised through proper effluent management (Powell et al. 2005; Gourley et al. 2012). The uncovered loafing pad modelled in our study resembles common facilities found in commercial dairy farms in Uruguay. These pads consist of a concrete feed pad and a bare ground bedding area, lacking drainage or effluent collection. They are used for cow confinement for silage supplementation during (substantial) periods of pasture shortage or extreme weather conditions (INALE 2019).

Effluent draining from such uncovered areas can significantly pollute ground and surface water due to high animal densities and manure accumulation (Houlbrooke et al. 2004). The stocking rate and time cows spend confined likely contribute to leaching losses from the loafing pad, as the amount of excreta deposited is closely tied to animal confinement duration (White et al. 2001). In our study, the systems had

high supplementation levels (Table 1), and cows were confined for a significant portion of time (GMax: 33% and GFix: 56% of the time; Stirling et al. 2021a). Although cows in the GFix strategy spent more time confined per year, N leaching in the off-grazing block was similar between feeding strategies, possibly because over the period with positive water balance (end autumn-early spring; Fig. 1), animals spent most of their time confined due to pasture shortage (data not presented).

The simulation analysis emphasises that in intensified hybrid dairy systems located in the subtropical region, addressing infrastructure limitations is crucial to mitigate N leaching and prevent water pollution (Aguerre et al. 2018; Fariña and Chilibroste 2019). Proper regulation and use of off-paddock facilities, such as stand-off or feed pads, are essential for sustainable production with higher stocking rates, protecting soils and cows, and reducing N leaching and emissions (de Klein 2001; Beukes et al. 2017).

The annual crop area significantly contributed to overall leaching losses in GFix systems with a higher proportion of maize and thus shorter-duration pastures in their pasture-crop sequence. Indeed, the second source of N leaching occurred within the annual crop block, specifically when the ground was left fallow. Annual crops, like maize, are often associated with large surpluses of soil mineral N after harvest and can lead to N losses through leaching (Betteridge et al. 2007; Beare et al. 2010). The combination of high levels of post-harvest soil mineral N and rainfall creates conditions favourable for increased drainage and N leaching losses below the root zone (Manevski et al. 2015). The model predicted that N leaching losses occurred during both fallow periods, before and after cultivation (Fig. 5a). This agrees with studies showing that fallow bare soil increases nitrate leaching risk as the opportunity for plant N uptake is removed (Mary et al. 1999; Cameron et al. 2013). Similar findings have been reported in non-grazed long-term maize cropping systems, with no significant losses during the growing season but higher rates during fallow periods (Tsimba et al. 2021). It needs to be noted that observed N leaching losses vary dramatically with the soil depth at which losses are measured; e.g., estimates from suction cups placed in a maize crop at a 700-mm depth were 3.5 times higher than those placed at a 1200-mm soil depth (Tsimba et al. 2021). Additionally, it has been reported that N

leaching rates in cropping systems increase with precipitation (Jabloun et al. 2015). In the modelled crop sequence, the soil was left fallow during the months with the highest rainfall, September–October and February–March, which accounted for 42% of the total annual rainfall (Fig. 1).

Several studies highlight the benefits of incorporating maize crops into dairy systems from productivity, agronomic, and environmental perspectives. However, when considering the land area used for maize silage production, the inclusion of maize crops in dairy systems may have a negative impact, leading to increased whole-farm N leaching losses (Ledgard et al. 2006; Basset-Mens et al. 2009; Beukes et al. 2017; Luo and Ledgard 2021). This is supported by various studies emphasising that the inclusion of input-intensive crops in the sequence, compared to pasture, not only raises N leaching but also increases energy use for operations as it requires the use of urea, other non-N-fertilisers, lime, and diesel for agricultural operations (Ledgard et al. 2006; Basset-Mens et al. 2009). To enhance the environmental performance of systems based on pasture-(double) crop sequences, strategies such as minimising fallow duration, and moving them to periods with negative water balance should be explored. This underscores the importance of tailoring effective mitigation strategies, specifically for maize cropping in Uruguay, and stresses the need for field measurements of soil N levels and losses to develop targeted approaches for reducing N leaching under local conditions.

Low emission intensity dominated by enteric CH_4

To perform a whole-farm assessment of the amount and sources of on-farm GHG emissions, we used a model based on the New Zealand GHG inventory (MFE 2018). An alternative approach could have been to use emission factors (EFs) based on Uruguay´s National Inventory (INGEI 2020). However, there is an absence of suitable whole-farm modelling tools to estimate GHG emissions based on Uruguay´s National Inventory. Therefore, to correctly interpret the results, it is important to consider the extent to which similarities and differences between EFs of the main sources of GHG emissions across countries would impact our conclusions.

Overseer''s EF for enteric fermentation is 96.5 kg CH_4 head^{-1} yr^{-1} (Tier 2, country-specific), while

87

Uruguay reports 108.4 kg CH4 head^{-1} yr^{-1} for dairy cattle (Tier 2, IPCC 2006). For N_2O emissions from excreta deposited in paddocks and effluent, the N excretion factor for dairy cattle used in Overseer is 112.8 kg N head^{-1} yr^{-1} (Tier 2, country-specific), compared to 92.5 kg N head^{-1} yr^{-1} as reported for Uruguay (Tier 2, IPCC 2006). Both countries employ an EF of 0.02 kg N_2O-N kg N^{-1} for excreta deposited into paddocks, following Tier 2, IPCC (2006) guidelines. Lastly, the emission factor for manure management is 0.0025 kg N_2O-N kg N^{-1} for New Zealand (Tier 2, country-specific) and 0.005 kg N_2O-N kg N^{-1} for Uruguay (Tier 2, IPCC 2006).

Regarding embodied CO_2 emissions (processing, transport, making on farm and feeding) from bought-in supplements, Overseer allocates an EF of 0.150 kg CO_2eq kg DM^{-1} (for user-defined concentrates) calculated based on life cycle analysis. There is no available information in Uruguay's dairy sector regarding EF of concentrates. Therefore, our analysis based on New Zealand´s EFs would underestimate CH_4 emissions (12%), an effect offset by the overestimation of N_2O emissions. Overall, these differences in EFs, which probably reflect a diet with lower nutritive value and lesser protein in Uruguayan dairy farms, do not alter the main inferences made in our study: low emission intensity dominated by CH_4.

Compared to commercial dairy farms in Uruguay (*c.f.* Darré et al. 2021), the four hybrid dairy systems assessed had higher total GHG emissions (~14,523 vs 4,491 kg CO_2e) and notably higher CO_2 emissions (~24% vs ~2% CO_2) because of greater use of concentrates and fertiliser. Primary GHG emissions from dairy farms include CH_4 and N_2O from enteric fermentation, manure storage and handling, and crop and pasture land (Rotz 2018). The results of the present study suggest that this is only partially true: in fact, N_2O was the minor contributor, and direct emission from applied fertilisers accounted for ~4% of total GHG emissions.

On the other hand, the assessed intensified hybrid systems had lower emissions per kg of fat protein corrected milk (~0.76 vs 1.09 kg CO_2e kg^{-1} FPCM; from Darré et al. 2021). Declines in emission intensity in response to intensification are common and often attributed to enhanced overall farm efficiency and production per cow and unit area (*e.g.* Christie et al. 2011; Lorenz et al. 2019; Fariña et al. 2024).

Total estimated GHG emissions are within the range reported for intensive pastoral systems in New Zealand (Ledgard et al. 2020; Luo and Ledgard 2021), Australia (Christie et al. 2011, 2018) and Ireland (O'Brien et al. 2012; Lahart et al. 2021). Compared to similar productivity, the hybrid systems evaluated in this study had lower GHG emissions per hectare than pasture-based systems in Ireland (Lahart et al. 2021) and New Zealand (Adler et al. 2015), as well as lower emissions per kilogram of milk solids (*c.f.* 12.5–14.0 kg CO_2e kg milk solids^{-1}). This would largely result from a higher consumption of energy-dense forages. Indeed, compared to year-round grazing systems with minimal supplementation, the intensified hybrid systems in this study had a profile with less CH_4 and more CO_2 emissions per hectare due to the use of energy-dense concentrates and maize silage (*c.f.* Adler et al. 2015; Ledgard et al. 2020; Lahart et al. 2021).

Interestingly, NZHF cows were predicted to emit less kg CO2e than NAHF cows (6.1 *vs.* 7.0 Mg CO_2e cow^{-1}, respectively), possibly because of their lower dry matter intake (18.6 *vs.* 20.3 kg DM cow^{-1} day^{-1}; Stirling et al. 2021a). However, farmlets with NZHF and NAHF cows had the same emissions per kg of milk solids because NZHF farmlets were stocked with more animals per ha (same live weight per ha), which agrees with previous studies on pasture-based systems comparing HF genotypes (O'Brien et al. 2010).

Conclusions

The conclusions drawn from this study serve as an initial identification of significant areas, time periods, and biochemical pathways of N losses and provide a quantitative profile of greenhouse gas sources.

Intensification strategies currently envisaged to maintain or increase the global competitivity of rainfed pasture-based dairy systems in humid subtropical regions, such as Uruguay, include feeding strategies featuring different pasture durations and proportions of annual winter-summer crops for silage (*e.g.* annual ryegrass-maize) within the pasture-crop sequence and cow genotypes with varying body sizes. These intensification paths resulted in increased total GHG emissions and significant N surpluses, *i.e.* N inputs not

converted into animal products and thus liable to be lost to the environment.

Nitrate leaching and ammonia volatilisation were identified as significant pathways for N loss, with feed/loafing pads identified as the critical hotspot. Regardless of the feeding strategy or cow genotype, addressing infrastructure limitations, especially effluent management in confinement areas, is crucial for mitigating impacts on water courses. Fallow periods in pasture-crop transitions emerged as the second most important source of N leaching. Therefore, systems with higher levels of home-grown maize silage in the diet would amplify environmental impacts by increasing fallow areas and excreta deposition in loafing pads.

The profile of GHG emissions revealed that fertiliser and N_2O emissions are comparatively minor contributors to total GHG emissions, which are amply dominated by emissions of enteric CH_4 and CO_2 generated by bought-in concentrates. Overall, the differences in EFs reported do not alter the main inferences made in our study: low emission intensity dominated by CH_4.

The conclusions emphasise the need for locally tailored mitigation approaches and field assessments to address the environmental repercussions of dairy intensification in humid subtropical climates. Insights gained on where critical environmental impacts reside provide a quantitative foundation for hypotheses to be empirically tested in future research on mitigation strategies. For instance, it appears that mitigation options should prioritise a deeper understanding of N cycling and synchronisation during transitions between annual crops and perennial pastures. These dynamics, in turn, contribute to identifying which sub-models of existing modelling tools must be given priority when local calibration is sought.

Declarations

References

Adler AA, Doole GJ, Romera AJ, Beukes PC (2015) Managing greenhouse gas emissions in two major dairy regions of New Zealand: A system-level evaluation. Agric Syst 135:1–9. https://doi.org/10.1016/j.agsy.2014.11.007

Aguerre M, Méndez MN, Torterolo S, Chilibroste P (2018) Pasture base dairy farm intensification: the role of growth strategy (stocking rate vs individual milk production) and availability of infrastructure. J Dairy Sci 101:255–256

Basset-Mens C, Ledgard S, Boyes M (2009) Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecol Econ 68(6):1615–1625. https://doi.org/10.1016/j.ecolecon.2007.11.017

Beare MH, Gillespie FTR, Maley S, Harrison-Kirk T, De Ruiter J (2010) Nitrate leaching from high production forage crop sequences. Proc New Zeal Grassl Assoc 72:23–28. https://doi.org/10.33584/jnzg.2010.72.2812

Becoña López G, Ledgard S, Wedderburn E (2013) A comparison of greenhouse gas emissions from Uruguayan and New Zealand beef systems. Agrocienc Urug 17(1):120–130. Available in: http://www.scielo.edu.uy/scielo.php?script=sci_arttext&pid=S2301-15482013000100015&lng=es&nrm=iso

Beretta-Blanco A, Carrasco-Letelier L (2021) Relevant factors in the eutrophication of the Uruguay River and the Río Negro. Sci Total Environ 761:143299. https://doi.org/10.1016/j.scitotenv.2020.143299

Betteridge K, Hoogendoorn CJ, Thorrold BS, Costall DA, Ledgard, SF, Park-Ng ZA, Theobald PW (2007) Nitrate leaching and productivity of some farming options in the Lake Taupo catchment. Proc New Zeal Grassl Assoc 69:123–129. https://doi.org/10.33584/jnzg.2007.69.2677

Beukes PC, Romera AJ, Gregorini P, Macdonald KA, Glassey CB, Shepherd MA (2017) The performance of an efficient dairy system using a combination of nitrogen leaching mitigation strategies in a variable climate. Sci Total Environ 599–600:1791–1801. https://doi.org/10.1016/j.scitotenv.2017.05.104

Bilotto F, Recavarren P, Vibart R, Machado CF (2019) Backgrounding strategy effects on farm productivity, profitability and greenhouse gas emissions of cow-calf systems in the Flooding Pampas of Argentina. Agric Syst 176:102688. https://doi.org/10.1016/j.agsy.2019.102688

Britt JH, Cushman RA, Dechow CD, Dobson H, Humblot P, Hutjens MF, Jones GA, Ruegg PS, Sheldon IM, Stevenson JS (2018) Invited review: Learning from the future—A vision for dairy farms and cows in 2067. J Dairy Sci 101:3722–3741. https://doi.org/10.3168/jds.2017-14025

Buckley C, Wall DP, Moran B, O'Neill S, Murphy PNC (2016) Farm gate level nitrogen balance and use efficiency changes post implementation of the EU nitrates Directive. Nutr Cycl Agroecosystems 104(1):1–13. https://doi.org/10.1007/s10705-015-9753-y

Burchill W, Lanigan GJ, Li D, Williams M, Humphreys J (2016) A system N balance for a pasture-based system of dairy production under moist maritime climatic conditions. Agric Ecosyst Environ 220:202–210. https://doi.org/10.1016/j.agee.2015.12.022

Cameron KC, Di HJ, Moir JL (2013) Nitrogen losses from the soil/plant system: A review. Ann Appl Biol 162(2):145–173. https://doi.org/10.1111/aab.12014

Chalar G, Garcia-Pesenti P, Silva-Pablo M, Perdomo C, Olivero V, Arocena R (2017) Weighting the impacts to stream water quality in small basins devoted to forage crops, dairy and beef cow production. Limnologica 65:76–84. https://doi.org/10.1016/j.limno.2017.06.002

Christie KM, Rawnsley RP, Eckard RJ (2011) A whole farm systems analysis of greenhouse gas emissions of 60 Tasmanian dairy farms. Anim Feed Sci Technol 166–167:653–662. https://doi.org/10.1016/j.anifeedsci.2011.04.046

Christie KM, Rawnsley RP, Phelps C, Eckard RJ (2018) Revised greenhouse-gas emissions from Australian dairy farms following application of updated methodology. Animal Production Sci 58:937–942. https://doi.org/10.1071/AN16286

Clark DA, Caradus JR, Monaghan RM, Sharp P, Thorrold BS (2007) Issues and options for future dairy farming in New Zealand. New Zeal J Agric Res 50:203–221. https://doi.org/10.1080/00288230709510291

Clay N, Garnett T, Lorimer J (2020) Dairy intensification: Drivers, impacts and alternatives. Ambio 49:35–48. https://doi.org/10.1007/s13280-019-01177-y

Darré E, Llanos E, Astigarraga L, Cadenazzi M, Picasso V (2021) Do pasture-based mixed dairy systems with higher milk production have lower environmental impacts? A Uruguayan case study. *New Zealand Journal of Agricultural Research* 64(3):444–462. https://doi.org/10.1080/00288233.2020.1750433

de Klein CAM (2001) An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. II. pasture production and cost/benefit analysis. New Zeal J Agric Res 44:217–235. https://doi.org/10.1080/00288233.2001.9513479

de Klein CAM, Monaghan RM, Ledgard SF, Shepherd, M (2010) A systems' perspective on the effectiveness of measures to mitigate the environmental impacts of nitrogen losses from pastoral dairy farming. In: Edwards GR, Bryant RH (eds) Meeting the challenges for pasture-based dairying. Proceedings of the 4th Australasian dairy science symposium, Christchurch, New Zealand. 31 Aug.– 2 Sept. 2010. Caxton Press, Caldwell, ID, pp 14–28

de Klein CAM, Monaghan RM, Gourley AM, CJP, Oenema O. Mark Powell J, (2017) Nitrogen performance indicators for dairy production systems. Soil Res 55:479–488. https://doi.org/10.1071/SR16349

Fariña S, Vigil Moreno O, Candioti F, Villanueva C, Sánchez Ledezma W, Moscoso CJ, Cajarville C, Charlón V, Urbina Abaunza L, GuacapiñaViteri A, Chirife S, Herrera D, Stirling S (2024) Milk production systems in Latin America and the Caribbean: biophysical, socioeconomic, and environmental performance. Agric Syst 218:103987. https://doi.org/10.1016/j.agsy.2024.103987

Fariña S, Chilibroste P (2019) Opportunities and challenges for the growth of milk production from pasture: the case of farm systems in Uruguay. Agric Syst 176:102631. https://doi.org/10.1016/j.agsy.2019.05.001

Fischer K, Burchill W, Lanigan GJ, Kaupenjohann M, Chambers BJ, Richards KG, Forrestal PJ (2016) Ammonia emissions from cattle dung, urine and urine with dicyandiamide in a temperate grassland. Soil Use Manag 32:83–91. https://doi.org/10.1111/sum.12203

Forrestal PJ, Krol DJ, Lanigan GJ, Jahangir MM, Richards KG (2017) An evaluation of urine patch simulation methods for nitrous oxide emission measurement. J Agric Sci 155(5):725–732. https://doi.org/10.1017/S0021859616000939

Gareli S, Mendoza A, Bello NM, Lattanzi FA, Fariña SR, Savian JV (2023) Effects of grazing management and concentrate supplementation on intake and milk production of dairy cows grazing orchardgrass. Anim Feed Sci Technol 301:115668. https://doi.org/10.1016/j.anifeedsci.2023.115668

Gorgoglione A, Gregorio J, Ríos A, Alonso J, Chreties C, Fossati M (2020) Influence of land use/land cover on surface-water quality of Santa Lucia River, Uruguay. Sustain 12(11):4692. https://doi.org/10.3390/su12114692

Gourley CJP, Aarons SR, Powell JM (2012) Nitrogen use efficiency and manure management in contrasting dairy production systems. Agriculture Ecosystem Environ 147:73–81. https://doi.org/10.1016/j.agee.2011.05.011

Harty MA, Forrestal PJ, Watson CJ, McGeough KL, Carolan R, Elliot C, Krol D, Laughlin RJ, Richards KG, Lanigan GJ (2016) Reducing nitrous oxide emissions by changing N fertiliser 812 use from calcium ammonium nitrate (CAN) to urea based formulations. Sci Total Environ 563:576–586. https://doi.org/10.1016/j.scitotenv.2016.04.120

Hyde BP, Forrestal PJ, Jahangir MM, Ryan M, Fanning AF, Carton OT, Lanigan GJ, Richards KG (2016) The interactive effects of fertiliser nitrogen with dung and urine on nitrous oxide emissions in grassland. Irish J Agricultural Food Res 55(1):1–9. https://www.jstor.org/stable/26194203

Hoekstra NJ, Schulte RPO, Forrestal PJ, Hennessy D, Krol DJ, Lanigan GJ, Müller C, Shalloo L, Wall DP, Richards KG (2020) Scenarios to limit environmental nitrogen losses from dairy expansion. Sci Total Environ 707:134606. https://doi.org/10.1016/j.scitotenv.2019.134606

Houlbrooke DJ, Horne DJ, Hedley MJ, Hanly JA, Snow VO (2004) A review of literature on the land treatment of farm-dairy effluent in New Zealand and its impact on

water quality. New Zeal J Agric Res 47(4):499–511. https://doi.org/10.1080/00288233.2004.9513617

INALE (2019) Encuesta Lechera 2019: resultados definitivos. Available in: https://www.inale.org/estadisticas/encuesta-lechera-2019-resultados-definitivos/. (In spanish)

INGEI (2020) Inventario nacional de gases de efecto invernadero 1990–2020 a la Conferencia de las Partes en la Convención Marco de las Naciones Unidas sobre el Cambio Climático. República Oriental del Uruguay. Available in: https://www.gub.uy/ministerio-ambiente/politicas-y-gestion/inventarios-nacionales-gases-efecto-invernadero-ingei. (In spanish)

IPCC (2006) 2006 IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K (eds) Published: IGES, Japan

Jabloun M, Schelde K, Tao F, Olesen JE (2015) Effect of temperature and precipitation on nitrate leaching from organic cereal cropping systems in Denmark. Eur J Agron 62:55–64. https://doi.org/10.1016/j.eja.2014.09.007

Jauregui JM, Michelini DF, Agnusdei MG, Baudracco J, Sevilla GH, Chilibroste P, Lattanzi FA (2017) Persistence of tall fescue in a subtropical environment: tiller survival over summer in response to flowering control and nitrogen supply. Grass Forage Sci 72:454–466. https://doi.org/10.1111/gfs.12252

Jauregui JM, Michelini DF, Sevilla GH, Berhongaray G, Berone GD, Baudracco J, Chilibroste P (2024) Tall fescue tiller survival over summer in a subtropical environment: the role of the size and depth of root systems. J Agron Crop Sci 210(1):e12682. https://doi.org/10.1111/jac.12682

Jones JW, Antle JM, Basso B, Boote KJ, Conant RT, Foster I, Godfray H, Herrero M, Howitt R, Janssen S, Keating B, Muñoz-Carpena R, Porter C, Rosenzweig C, Wheeler TR (2017) Toward a new generation of agricultural system data, models, and knowledge products: State of agricultural systems science. Agric Syst 155:269–288. https://doi.org/10.1016/j.agsy.2016.09.021

Kelly P, Shalloo L, Wallace M, Dillon P (2020) The Irish dairy industry – Recent history and strategy, current and future challenges. Int J Dairy Technol 73:309–323. https://doi.org/10.1111/1471-0307.12682

Kruk C, Segura A, Piñeiro G, Baldassini P, Pérez-Becoña L, García-Rodríguez F, Perera G, Piccini C (2023) Rise of toxic cyanobacterial blooms is promoted by agricultural intensification in the basin of a large subtropical river of South America. Glob Change Biol 29:1774–1790. https://doi.org/10.1111/gcb.16587

Lahart B, Shalloo L, Herron J, O'Brien D, Fitzgerald R, Boland TM, Buckley F (2021) Greenhouse gas emissions and nitrogen efficiency of dairy cows of divergent economic breeding index under seasonal pasture-based management. J Dairy Sci 104(7):8039–8049. https://doi.org/10.3168/jds.2020-19618

Lazzarini B, Baudracco J, Tuñon G, Gastaldi L, Lyons N, Quattrochi H, Lopez- Villalobos N (2019) Milk production from dairy cows in Argentina: current state and perspectives for the future. Appl Anim Sci 35(4):426–432. https://doi.org/10.15232/aas.2019-01842

Ledgard SF, Falconer SJ, Abercrombie R, Philip G, Hill JP (2020) Temporal, spatial, and management variability in the carbon footprint of New Zealand milk. J Dairy Sci 103:1031–1046. https://doi.org/10.3168/jds.2019-17182

Ledgard SF, Sprosen M, Judge A, Lindsey S, Jensen R, Clark D, Luo J (2006) Nitrogen leaching as affected by dairy intensification and mitigation practices in the resources-efficient dairying (RED) trial. Curry L, Hanley J (eds) Implementing sustainable nutrient management strategies in agriculture, Massey University Press, Palmerston North, New Zealand, pp 263–268

Leip A, Bodirsky BL, Kugelberg S (2021) The role of nitrogen in achieving sustainable food systems for healthy diets. Glob Food Sec 28:100408. https://doi.org/10.1016/j.gfs.2020.100408

Lorenz H, Reinsch T, Hess S, Taube F (2019) Is low-input dairy farming more climate friendly? A meta-analysis of the carbon footprints of different production systems. J Clean Prod 211:161–170. https://doi.org/10.1016/j.jclepro.2018.11.113

Luo J, Ledgard S (2021) New Zealand dairy farm systems and key environmental effects. Front Agric Sci Eng 8:148–158. https://doi.org/10.15302/J-FASE-2020372

Manevski K, Børgesen CD, Andersen MN, Kristensen IS (2015) Reduced nitrogen leaching by intercropping maize with red fescue on sandy soils in North Europe: a combined field and modeling study. Plant Soil 388:67–85. https://doi.org/10.1007/s11104-014-2311-6

Mary B, Beaudoin N, Justes E, Machet JM (1999) Calculation of nitrogen mineralisation and leaching in fallow soil using a simple dynamic model. Eur J Soil Sci 50:549–566. https://doi.org/10.1046/j.1365-2389.1999.00264.x

McDowell RW, Rotz CA, Macintosh OJ, KA, (2022) Limiting grazing periods combined with proper housing can reduce nutrient losses from dairy systems. Nature Food 3(12):1065–1074. https://doi.org/10.1038/s43016-022-00644-2

MFE (2018) Ministry for the environment. New Zealand's greenhouse gas inventory 1990–2016. In: Fulfilling Reporting Requirements under the United Nations Framework Convention on Climate Change and the Kyoto Protocol. New Zealand Government. Available in: https://environment.govt.nz/assets/Publications/New-Zealands-Greenhouse-Gas-Inventory-1990-2019-Volume-1-Chapters-1-15.pdf

Morales-Piñeyrúa JT, Damián JP, Banchero G, SantAnna AC (2022) The effects of heat stress on milk production and the grazing behavior of dairy Holstein cows milked by an automatic milking system. J Animal Sci 100(9):skac225. https://doi.org/10.1093/jas/skac225

O'Brien D, Shalloo L, Grainger C, Buckley F, Horan B, Wallace M (2010) The influence of strain of Holstein-Friesian cow and feeding system on greenhouse gas emissions from pastoral dairy farms. J Dairy Sci 93(7):3390–3402

O'Brien D, Shalloo L, Patton J, Buckley F, Grainger C, Wallace M (2012) Evaluation of the effect of accounting method, IPCC v. LCA, on grass-based and confinement dairy systems' greenhouse gas emissions. Animal 6(9):1512–1527. https://doi.org/10.1017/S1751731112000316

OECD/FAO (2021) OECD-FAO agricultural outlook 2021–2030, OECD Publishing, Paris. https://doi.org/10.1787/19428846-en

Oenema J, Oenema O (2021) Intensification of grassland-based dairy production and its impacts on land, nitrogen and phosphorus use efficiencies. Front Agric Sci Eng 8(1):130–147. https://doi.org/10.15302/J-FASE-2020376

Ojeda JJ, Pembleton KG, Caviglia OP, Islam MR, Agnusdei MG, Garcia SC (2018a) Modelling forage yield and water productivity of continuous crop sequences in the Argentinian Pampas. Eur J Agron 92:84–96. https://doi.org/10.1016/j.eja.2017.10.004

Ojeda JJ, Caviglia OP, Agnusdei MG, Errecart PM (2018b) Forage yield, water- and solar radiation-productivities of perennial pastures and annual crops sequences in the south-eastern Pampas of Argentina. Field Crop Res 221:19–31. https://doi.org/10.1016/j.fcr.2018.02.010

Ortega G, Berberian N, Chilibroste P (2024) The effects of stocking rate residual sward height and forage supplementation on forage production feeding strategies and productivity of milking dairy cows. Front Anim Sci 5:1319150. https://doi.org/10.3389/fanim.2024.1319150

Powell JM, McCrory D, Jackson-Smith D, Saam H (2005) Manure collection and distribution on Wisconsin dairy farms. J Environ Qual 34:2036–2044. https://doi.org/10.2134/jeq2004.0478

Powell JM, Gourley CJP, Rotz CA, Weaver DM (2010) Nitrogen use efficiency: A potential performance indicator and policy tool for dairy farms. Environ Sci Policy 13(3):217–228. https://doi.org/10.1016/j.envsci.2010.03.007

Rawnsley RP, Langworthy AD, Pembleton KG, Turner LR, Corkrey R, Donaghy DJ (2014) Quantifying the interactions between grazing interval, grazing intensity, and nitrogen on the yield and growth rate of dryland and irrigated perennial ryegrass. Crop Pasture Sci 65(8):735–746. https://doi.org/10.1071/CP13453

Roche L, Forrestal PJ, Lanigan GJ, Richards KG, Shaw LJ, Wall DP (2016) Impact of fertiliser nitrogen formulation and N stabilisers on nitrous oxide emissions in spring barley. Agriculture Ecosystems & Environment 233:229–237. https://doi.org/10.1016/j.agee.2016.08.031

Roche JR, Berry DP, Bryant AM, Burke CR, Butler ST, Dillon PG, Donaghy DJ, Horan B, Macdonald KA, Macmillan KL (2017) A 100-Year Review: A century of change in temperate grazing dairy systems. J Dairy Sci 100(12):10189–10233. https://doi.org/10.3168/jds.2017-13182

Rotz CA (2018) Symposium review: Modeling greenhouse gas emissions from dairy farms. J Dairy Sci 101(7):6675–6690. https://doi.org/10.3168/jds.2017-13272

Rotz CA, Holly M, de Long A, Egan F, Kleinman PJA (2020) An environmental assessment of grass-based dairy production in the northeastern United States. Agric Syst 184:102887. https://doi.org/10.1016/j.agsy.2020.102887

ROU (2008) Decreto 405/008. Uso responsable y sostenible de los suelos. República Oriental del Uruguay (ROU). Available in: https://www.impo.com.uy/bases/decretos-originales/405-2008. (In Spanish)

Rowarth JS, Parsons AJ (2015) Rethinking production systems: science for the land-based sector. Journal of the Royal Society of New Zealand 45(2):108–113. https://doi.org/10.1080/03036758.2015.1011173

Scheneiter JO, Bertín OD, Lattanzi FA (2019) Under stressful conditions, high cattle density maintains white clover abundance. Agronomy Journal 111(3):1198–1207. https://doi.org/10.2134/agronj2018.03.0186

Selbie DR, Watkins NL, Wheeler DM, Shepherd M (2013) Understanding the distribution and fate of nitrogen and phosphorus in OVERSEER®. Proc New Zeal Grassl Assoc 75:113–118. https://doi.org/10.33584/jnzg.2013.75.2931

Stirling S, Delaby L, Mendoza A, Fariña S (2021a) Intensification strategies for temperate hot summer grazing dairy systems in South America: Effects of feeding strategy and cow genotype. J Dairy Sci 104(12):12647–12663. https://doi.org/10.3168/jds.2021-20507

Stirling S, Fariña S, Pacheco D, Vibart R (2021b) Whole-farm modelling of grazing dairy systems in Uruguay. Agric Syst 193:103227. https://doi.org/10.1016/j.agsy.2021.103227

The JAMOVI project (2023) JAMOVI (Version 2.3) [Computer Software]. Retrieved from https://www.jamovi.org

Tsimba R, Gunn T, Densley R, Williams I, Edmeades G, Millar J (2021) Quantification and mitigation of nitrogen leaching in a maize silage cropping system. J New Zealand Grasslands 83:163–170. https://doi.org/10.33584/jnzg.2021.83.3506

Wachendorf C, Taube F, Wachendorf M (2005) Nitrogen leaching from 15 N labelled cow urine and dung applied to grassland on a sandy soil. Nutr Cycl Agroecosyst 73:89–100. https://doi.org/10.1007/s10705-005-8313-2

Wheeler DM, Ledgard SF, de Klein CA (2008) Using the OVERSEER nutrient budget model to estimate on-farm greenhouse gas emissions. Aust J Exp Agric 48:99–103. https://doi.org/10.1071/EA07250

Wheeler DM, Ledgard SF, de Klein CA, Monaghan RM, Carey PL, McDowell R W, Johns KL (2003) OVERSEER® nutrient budgets-moving towards on-farm resource accounting. Proc New Zeal Grassl Assoc 65:191–194. https://doi.org/10.33584/jnzg.2003.65.2484

White SL, Sheffield RE, Washburn SP, King LD, Green JT (2001) Spatial and time distribution of dairy cattle excreta in an intensive pasture system. J Environ Qual 30(6):2180–2187. https://doi.org/10.2134/jeq2001.2180