Co-benefits and trade-offs of water quality mitigation measures on greenhouse gas emissions from New Zealand dairy systems

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Abstract

As part of government climate change policy, New Zealand dairy farmers will be encouraged to reduce their greenhouse gas (GHG) emissions through a proposed pricing mechanism. With integrated farm plans on the horizon, farmers need information on how mitigations for water quality will impact GHG emissions. Using a typology approach that captured the main production attributes and drivers of nitrogen (N) and phosphorus (P) losses to water we assessed the impact of 12 current N and P contaminant mitigations on GHG co-benefits or trade-offs. Four of the mitigations had a co-benefit effect, with most of these being N mitigation measures. Trade-offs were detected for two water quality mitigations (stand-off pads and deferring effluent application), resulting in an increase in estimated GHG emissions. The remaining six water quality mitigations tested, either had a minimal impact, or had both a trade-off and co-benefit. Our data provides pastoral farmers and rural professionals with information to guide initial conversations on options to reduce losses to water and air for developing integrated farm plans.

Key words; water quality, dairy typologies, methane, nitrous oxide, carbon dioxide, dairy intensity

Introduction

The ability of farmers to mitigate losses to both water and air is becoming increasingly important. The current state of water quality in New Zealand rivers and lakes shows that the health of 46% of lakes can be considered poor and 45% of these continue to show declining health (Ministry for the Environment and Stats NZ 2023). In addition, 19% of monitored groundwater sites have nitrate levels above the national drinking water standard of 11.3 g/m^3 (Ministry for the Environment and Stats NZ 2023). Furthermore, the agricultural sector is the largest source of greenhouse gas (GHG) emissions in New Zealand, contributing 49% of the total (Ministry for the Environment 2023). The dairy sector represents the largest contributor, accounting for 48% of agricultural emissions, of which 73% is derived from enteric methane (CH_4), 8% from manure management (mainly CH_4 from methanogenic activity in effluent storage ponds) and 17% from nitrous oxide (N_2O) emitted from N inputs such as urine, dung and N fertiliser. Nitrous oxide emissions can occur directly from soils (amounting to 14% of the 17% of N_2O emissions), primarily due to nitrification and denitrification soil processes (Butterbach-Bahl et al. 2013), and indirectly, following re-deposition of volatilised ammonia and N_2O emissions from water bodies containing leached N, together representing the remaining 3% of the N_2O emissions. The remaining 2% of GHG emissions from dairy farms is represented by carbon dioxide (CO_2) emitted from the dissolution of urea fertiliser following application to land (Ministry for the Environment 2023). As part of the government's commitment to tackling climate change, New Zealand has adopted GHG reduction targets of net zero emissions of N_2O and urea fertiliser-induced CO_2 emissions, and 27 to 45% reductions in methane (CH_4) emissions by 2050 (Office of the Minister for Climate Change 2019).

There has been considerable effort to reduce N and P losses to water. A summary of such mitigation strategies to date, including information on their effectiveness and uptake by farmers, was undertaken by Monaghan et al. (2021b). They showed that P loss mitigations have generally been successful in mitigating P loss to water. The effectiveness and uptake of N mitigations, however, have not been sufficient to counter the increase in dairy intensification. McDowell et al. (2020) suggested that the full implementation of known mitigations to dairy farms could see a further decrease in nutrient loss to waterways of 34% and 26% for N and P respectively.

More recently, He Waka Eke Noa, an agricultural sector, Māori and Government partnership proposed a system to price GHG emissions from agricultural activities as an alternative to the New Zealand Emissions Trading Scheme (NZ ETS) (He Waka Eke Noa, 2022). It is likely to see New Zealand agricultural producers, including dairy farmers, encouraged to

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reduce their GHG emissions or pay through some type of pricing mechanism.

Previous workers investigating mitigation strategies for GHGs have indicated that reduced N fertiliser inputs, coupled with reduced stock intensity showed the greatest promise for lowering GHG emissions (Doole 2014; Adler et al. 2013, 2015). Similarly, Beukes at al. (2019) found that reducing N fertiliser inputs and purchased feed, with an associated stocking rate reduction, resulted in lower N leaching and lower GHG emissions. The reduction in CH_4 emissions was primarily driven by a reduction in the amount of feed produced and therefore offered to livestock, together with a reduction in the stocking rate. Nitrous oxide emissions were also reduced by less feed and therefore fewer animals, but also through lower dietary N content, which can be achieved by replacing high N content supplements with those containing lower N concentrations (van der Weerden et al. 2018). However, Beukes et al (2010) suggested that lowering stocking rate, without lowering production, requires efficiency gains in both cow production through genetics and pasture management. A study by Vibart et al. (2015) in Southland, New Zealand found that GHG abatement was possible with nutrient loss mitigation strategies on dairy farms and the greater the farm intensity the greater the GHG abatement.

While farmers are already adopting many of the mitigations targeting nutrient losses to water (Monaghan 2021b), they are also having to develop farm plans and understand the implications of such mitigations for GHG emissions as part of the government's climate action plan and the response presented by He Waka Eke Noa (HWEN) (Selbie et al. 2021). With pastoral farmers facing the challenge of mitigating losses to both water and air, there is a need for improved understanding of co-benefits and trade-offs associated with mitigation measures, so that future mitigation actions taken on farm are cost-effective.

This paper aims to quantify how mitigation strategies targeting N and P loss to water may impact dairy farm greenhouse gas (GHG) footprints in terms of co-benefits or trade-offs over a range of dairy typologies and intensities (i.e. levels of stocking rate and supplement usage). A typology approach captures the effects of climate, topography and soil properties on the inherent vulnerability of contaminant loss risk (Monaghan et al. 2021a). Five groups of mitigation strategies were evaluated: (i) improved N management, (ii) utilising off-paddock cow facilities, (iii) improved effluent management, (iv) edge of field attenuation (i.e. constructed wetlands) and (v) improved irrigation management.

Previous research on co-benefits and trade-offs provides an indication of direction of travel. For instance, improved N management provides co-benefits for reducing GHG emissions through reduced feed production and lower dietary N (N management) (e.g. de Klein et al 2020; van der Weerden et al. 2018), while improved irrigation management will result in lower indirect N_2O emissions through reduced N leaching (Vogeler et al. 2019). In contrast, inclusion of off-paddock cow facilities can have a trade-off effect on GHG emissions, due to increased emissions from a larger volume of stored manure offsetting any reduction in N_2O emissions from less urine and dung deposition on soil (e.g. van der Weerden et al. 2018). Similarly, improved effluent management, such as deferred effluent application for reducing contaminant losses to water will require longer periods of effluent storage, which can increase manure-derived CH_4 emissions (Laubach et al. 2015). For some mitigation measures there is limited information on the trade-offs and co-benefits. One such example is inclusion of edge of field attenuation via constructed wetlands, which may have a co-benefit through lower indirect N_2O emissions via reduced N leaching, although data is limited (Simon et al. 2021). Similarly, avoiding N fertiliser from effluent blocks is likely to reduce N_2O emissions due to reduced fertiliser inputs (Hinton et al. 2015).

Based on this previous research, we hypothesise that improved N management, edge of field attenuation (i.e. constructed wetlands) and improved irrigation management will provide co-benefits for reducing GHG emissions while utilising off-paddock cow facilities will result in a trade-off with increased GHG emissions. Lastly, we suggest that improved effluent management will provide either co-benefits or trade-offs, depending on the specific mitigation measure.

Our study will test these hypotheses and provide pastoral farmers and rural professionals with the type of information required for an initial conversation on options to reduce contaminants losses, from which more informed decisions can be investigated for individual farms.

Materials and methods

We developed a farm typology framework, building on the typologies previously developed by Monaghan et al. (2021a). In this previous study farms representing each typology were constructed based on mean values for area and herd size, and associated settings for farm inputs, management practices and production information. The use of such a framework allows for modelled losses and mitigation effectiveness to be assessed at scales where measured data may not be available (Srinivasan et al. 2021).

Typology selection

A hierarchical analysis of three primary attributes that

Table 1 Characteristics of the typologies together with the estimated areas and numbers of DairyBase farm files used to generate typical farms covering three intensities for each typology.

Typology #	Soil drainage	Soil wetness	Slope	Area of dairy farms (ha)	Number of files 2014-2015	DairyBase 2019-2020
1	light soil	Irrigated	Flat	225076	14	20
2	light soil	Dry	Flat	137146	11	11
3	light soil	Irrigated	Rolling	529	0	0
4	light soil	Dry	Rolling	12349	0	0
5	light soil	Moist	Flat	293439	24	35
6	light soil	Moist	Rolling	17768	1	2
7	light soil	Wet	Flat	84020	5	10
8	light soil	Wet	Rolling	19909	0	1
9	poorly drained	Irrigated	Flat	61984	7	5
10	poorly drained	Dry	Flat	176225	13	9
11	poorly drained	Irrigated	Rolling			
12	poorly drained	Dry	Rolling	3838	1	0
13	poorly drained	Moist	Flat	195160	0	32
14	poorly drained	Moist	Rolling	10876	0	0
15	poorly drained	Wet	Flat	51708	1	1
16	poorly drained	Wet	Rolling	3795	0	0
17	well drained	Irrigated	Flat	111046	7	9
18	well drained	Dry	Flat	126666	10	11
19	well drained	Irrigated	Rolling	696	0	0
20	well drained	Dry	Rolling	4050	1	0
21	well drained	Moist	Flat	622378	47	69
22	well drained	Moist	Rolling	41700	0	2
23	well drained	Wet	Flat	173357	3	21
24	well drained	Wet	Rolling	15585	1	2

influence contaminant losses to water was used to classify farms into discrete typologies that represent contrasting levels of contaminant loss risk:

1. **Wetness** was used to categorise variability in contaminant loss due to the influence of water availability and transport. Four classes of wetness were distinguished based on the following criteria:

(1a) 'Irrigated' – farms where >50% of the farm area is irrigated.

(1b) 'Wet' – farms where mean annual rainfall exceeded 1700 mm.

(1c) 'Moist' – farms where mean annual rainfall was between 1100 mm and 1700 mm.

(1d) 'Dry' – farms where mean annual rainfall was less than 1100 mm.

2. **Soil Drainage** was used to capture the effects of soil processes that influence the vulnerability of soil to nitrate leaching. Soil drainage class, as defined in the Land Resource Information Systems (LRIS) soil map layers (Newsome et al. 2008), was selected as the attribute that best represented these aspects of N leaching vulnerability. Three classes of soil drainage were thus determined:

(2a) 'Light' soils, defined as having plant available water holding capacity to 60 cm (PAW60cm) of less than 85 mm.

(2b) 'Well-drained' soils, classified as 'well' or 'moderately well' drained in the LRIS mapping system; and

(2c) 'Poorly-drained' soils, classified as having 'imperfect', 'poor' or 'very poor' soil drainage classes (Newsome et al. 2008).

3. **Slope** was used as the third attribute considered to influence contaminant losses to water. Two slope classes were distinguished, either Flat/Undulating (0–7°) or Rolling (7–15°), primarily to reflect vulnerability to P runoff (McDowell et al. 2005).

Using the typology attributes (wetness, drainage and slope), a total of 24 dairy farm typologies were identified covering the four wetness classes, three drainage classes and two slope classes as described in Table 1.One of these typologies (number 11) had no dairy farms present so was discarded. A further two typologies (Nos. 3 and 19) only had small areas of dairying and were similar in that they were both rolling and irrigated but differed by soil drainage (light and well-drained) so they were combined for the purposes of our study, resulting in 22 typologies.

Management systems within typologies

We included three levels of farm management **intensity** within each typology to provide a range of outputs considered as representative of each respective typology. Three Overseer files were created for each typology: a base file representing a typical average farm for that typology, one representing a higher intensity farm and one representing a lower intensity farm. Where sufficient observations from DairyBase existed (i.e. >5), we took a statistical approach to identifying the low and high intensity farm systems by adopting the 25th and 75th percentile of the different management inputs. For typologies where there were insufficient observations for applying this statistical approach (i.e. ≤5) , we applied an approximation of the variation in intensity. The management inputs used for the different intensities were designed to fit within the DairyNZ farm production system definitions (DairyNZ 2021). For the majority of typologies, the three intensities equated to farm systems 2, 3, and 4. For four typologies the lowest intensity system equated to a system 1,, while the highest intensity system equated to a system 5 for 3 typologies.

The management inputs were changed, where appropriate, to reflect the differing management intensities, based on the following reasoning:

a. **Stocking rate** (SR). The base farm stocking rate was set to the mean of the DairyBase files. We found that the 25th and 75th percentile represented a stocking rate variation of approximately +/- 0.5 cows/ha. Therefore, for typologies where there were insufficient observations for applying our statistical approach (i.e. less than 5), we applied a +/- 0.5 cows/ha stocking rate variation.
b. **Fertiliser N inputs**. As with SR, fertiliser N for the base farm was set at the mean of the supplied farm data. The high and low intensity N inputs were also set at the 75th and 25th quartiles, with the maximum N fertiliser limited to 190 kg N/ha to meet regulatory requirements (Ministry for the Environment 2021). On those typologies where the range of N fertiliser inputs was low, or all above 190 kg N/ha, one N fertiliser rate was used across the three farm intensities, with a maximum of 190 kg N/ ha. Where the N fertiliser rate was lowered, it was assumed that pasture production would be lowered by a calculated amount using a N fertiliser response rate derived from the farm location, expected pasture growth rates and timing of N applications (DairyNZ 2021). Stocking rates were adjusted accordingly assuming a minimum pasture intake of 4500 kg DM/cow/year (Romera and Doole 2015).
c. **Milk solids production**. This was set to the mean milk solids production (kg MS/cow) of the supplied DairyBase farm data for all three intensities within each typology.
d. **Supplements.** Imported supplements brought on to the base (typical) farm were set to the mean of that from the supplied DairyBase farm data. Types of supplements used were those contained in the *Overseer* files used for Monaghan et al. (2021a). For the low and high intensity farms, pasture production was adjusted based on changes to N fertiliser application alone, with the amount of supplement imported then modified to meet the remaining energy requirements of the milking herd.

Losses to water and air

Farm nutrient budgets used previously by Monaghan et al. (2021a), were updated to the latest version of the Overseer® Nutrient Budgeting software (version 6.5.0) (Wheeler et al. 2008, 2011; Cichota et al. 2012), hereafter referred to as *Overseer*. In addition, certain management and production inputs were updated to better represent current farm systems for the 2019-2020 production year using data from DairyNZ DairyBase baseline farm records (Table 1). Where existing *Overseer* files were unavailable, new *Overseer* nutrient budget files were generated using the climate, topography and soil properties typical for that typology. Farm management inputs were obtained from the DairyBase files where these were available.

Nitrogen losses (kg N/ha/year) modelled by *Overseer* account for N run-off and leaching below the root zone from both livestock urine patches and from other non-urine sources such as fertiliser, or soil mineralisation. Phosphorus losses (kg P/ha/year) modelled by *Overseer* relate to both overland runoff (main loss process) and leaching below the root zone.

Greenhouse gas emissions are split into short-lived gases (CH_4) and long-lived GHG (N_2O and carbon dioxide (CO_2)), thereby aligning with the suggested methodology for domestic reporting of farm-scale GHG emissions (He Waka Eke Noa, 2022). Methane emissions are reported as kg CH_4 /ha/year, whereas both long-lived gases are reported as CO_2 equivalents (kg CO_2e /ha/year), where N_2O is converted to equivalent CO_2 emissions using the 100-year time horizon global warming potential of 298 kg CO_2-equivalent per kg N_2O (Forster et al. 2007). Long-lived gases N_2O and CO_2 are reported separately to aid interpretation of drivers of co-benefits and trade-offs. Sources of GHG emissions are limited to the following (He Waka Eke

Table 2 Mitigation measures applied to dairy farms to assess farm scale losses of N and P to water and changes to GHG emissions.

Management strategy	Mitigation measure	Details	Alignment to typology structure
Nitrogen management	Judicious scheduling of N fertiliser to avoid risk months	No N fertiliser applied May, June or July	All
	Reducing excessive inputs of N in fertiliser and feed*	Fertiliser N reduced by 20%, and replace PKE (high N supplement) with Maize or cereal grain /silage (low N supplement)	All
	No N fertiliser applied to pastures*	Animal numbers and production adjusted accordingly	All
Off-paddock cow facilities	Use of a standoff pad (sawdust, bark and woodchip base) for May, June, July and August. Manure stored in heap for 5 months prior to land application.	50% of animals off pasture on pad for 18 hours/day May and August 100% of animals off pasture or crop on pad for 18 hours/day June and July	All
	Uncovered wintering pad (sawdust, bark and woodchip base). Manure stored in heap for 5 months prior to land application.	100 % of animals off pasture (15 typologies) or crop (7 Typologies) on pad for June and July, 50% in August, for 24 hours/day	All
	Covered wintering barn (concrete base, regularly scraped). Slurry stored uncovered for 5 months prior to land application.	100 % of animals off pasture (15 typologies) or crop (7 Typologies) in barn for June and July, 50% in August for 24 hours/day	All
Effluent management	Enlarged areas receiving effluent	Effluent area enlarged so that K inputs in effluent, fertiliser and supplements were less than 75 kg K/ha/yr	All
	Targeting N fertiliser applications to non-effluent areas*	No N fertiliser applied to effluent areas	All
	Deferred effluent application	No effluent applied in May, June or July	All
	Low-rate effluent application	Effluent applied via low-rate application and actively managed to avoid overland flow or ponding	All
Edge of field attenuation	Artificial wetland	Artificial wetland added to appropriate typologies. The wetland covered 2% of the catchment area which was set to 80% of the milking platform (Tanner and Kadlec 2013)	Poorly drained soils and farms on rolling slopes
Irrigation management 1	Reduced overwatering by adjusting application depth via sensors to meet deficit target.	Irrigation management was by means of soil moisture sensors with irrigation scheduled at 50% to 70% of PAW depending on soil type. Irrigation applied to achieve a soil moisture target of 95% PAW.	Irrigated farms where irrigation is applied at a fixed depth and fixed timing
Irrigation management 2	Reduced overwatering by adjusting application depth via sensors to meet deficit target.	Irrigation management was by means of soil moisture sensors with irrigation scheduled at 50% to 70% of PAW depending on soil type. Irrigation applied to achieve a soil moisture target of 95% PAW.	Irrigated farms where irrigation scheduling is done via a water balance model.

* Pasture production was reduced due to mitigation measure.

Noa, 2022), with all upstream emissions (e.g. from transport, electricity, CO_2 from fertiliser manufacturing, etc) excluded:

- CH_4 from enteric fermentation and manure management
- N_2O from manure management and N fertiliser use, including indirect emissions
- CO_2 from urea fertiliser dissolution

Mitigation modelling

Mitigations used to reduce N and P losses to water were those previously recorded (Monaghan et al. 2021a, b) as being already accepted and implemented or deemed as developing or likely to be partially or fully implemented in future (McDowell et al. 2020). Details of these are given in Table 2. An additional mitigation of removing N fertiliser from the pastoral farming system (zero N) was also included.

Mitigations were modelled by changing the appropriate management settings within *Overseer* for each of the farm systems within each typology. Results from the irrigation mitigation were compared with: (1) base farms where irrigation application management was minimal (i.e. set depth and irrigation return period) and (2) base farms where some improved irrigation management was already used via application scheduling using a water balance model (i.e. application depth adjusted to meet requirements with a fixed return period).

Data management

Estimations of N and P losses from the root zone, and CH_4, N_2O and CO_2 emissions for the 66 base farms (22 typologies x 3 intensities) were tabulated. Mitigation effectiveness was then calculated as the percentage change of the applied mitigation compared to the base farm losses, where positive values indicate a reduction in loss and negative values indicate an increase in loss.

An ANOVA analysis was carried out on the base farm losses using GENSTAT (v21) to ensure that typologies were discretely different in their losses. A regression analysis was then completed to indicate the order of importance of the typology attributes: moisture, soil drainage and slope as well as that of farm intensity for each water contaminant and GHG type. For this analysis we used the DairyNZ farm system numbers 1-5 to represent farm intensity, to ensure all data from the 66 base farms (22 typologies x 3 intensities) could be appropriately compared. The sum of squares represent the portion of the total variation in the data set that can be attributed to a particular variable, therefore are presented to differentiate the relative importance of the typology attributes and management intensity for both N and P losses to water and the three forms of GHG losses.

Results

Base farms

There was a wide range of estimates of N and P loss to water from the 22 typologies modelled (Table 3). For N losses, modelled estimates ranged from 17 to 141 kg N/ha. Soil drainage status and soil wetness were the main typology attributes influencing estimates of N loss to water (Table 4). Management intensity and slope were of lesser importance, having relatively minor effects on estimates of N loss. Estimates of P loss ranged from 0.5 to 6.4 kg P/ha, with soil wetness being the most important attribute, closely followed by soil drainage class and slope. For P loss, the interactions between soil drainage, soil wetness and slope were of similar importance to the main attributes (data not shown). The highest losses of P occurred on the poorly drained, rolling typology for both the moist and wet soils. Farm intensity had no impact on estimates of P loss.

For the estimates of CH_4 emissions the values ranged from 198 to 495 kg CH_4/ ha. The key attribute influencing CH_4 was farm intensity as indicated by feed offered per ha (Figure 1A), followed by soil wetness and drainage. Slope only had a very small impact on estimates of CH_4 emissions (Table 4). Nitrous oxide emissions ranged from 1333 to 3498 kg CO_2e/ha, with farm intensity as the main driver of emissions (Figure 1B), followed by soil wetness and drainage. Carbon dioxide emissions from urea fertiliser dissolution following application ranged from 177 to 770 kg CO_2e/ha. The main driver of CO_2 emissions was soil wetness, followed closely by farm intensity (Figure 1C) then drainage and slope (Table 4).

Mitigation effectiveness

The effects of individual mitigation measures on N or P losses to water and GHG emissions across the 22 typologies are shown in Figures 2 (N management), 3 (off paddock cow facilities), 4 (effluent management) and 5 (wetlands and irrigation management). Mitigations highlighted in boxes indicate co-benefits or trade-offs with GHG reductions, respectively. Preventing the application of N fertiliser over the months of May – July generally reduced N losses to water, ranging from -2% to 13% reduction (where a negative value corresponds to an increase in loss), with no effect of P losses. There was no or minimal co-benefit or trade-off effect on GHG emissions (Figure 2A). Reducing N inputs in feed and fertiliser by 20% resulted in a reduction in N leaching of 2 to 25% and P losses of 0 to 13%. This mitigation measure generally

Table 3 Base farm estimated pasture production and losses of N and P to waterways and emissions of CH4, N2O and urea CO2 for dairy farm typologies grouped by moisture, drainage, slope, and farm intensity (Low or High)

Moisture	Typ #	Drainage	Slope	Pasture production (t DM/ha)		N loss (kg N/ha)		P loss (kg N/ha)		CH_4 (kg CH_4/ha)		N_2O (kg CO_2e/ha)		Urea CO_2 (kg CO_2e/ha)	
				Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Dry	2	Light soil	Flat	14.2	16.1	50	72	0.6	0.7	280	387	2156	2986	273	681
	4	Light soil	Rolling	12.3	13.4	49	63	0.6	0.7	256	363	1936	2549	425	770
	18	Well drained	Flat	14.8	16.3	23	34	0.5	0.8	291	384	2044	2679	295	751
	20	Well drained	Rolling	11.0	13.0	22	30	0.5	0.6	251	365	1869	2578	437	756
	10	Poorly drained	Flat	14.4	16.7	17	27	1.4	1.5	287	412	2084	2973	287	728
	12	Poorly drained	Rolling	12.2	13.1	21	25	1.2	1.4	320	416	2226	2682	410	609
Moist	5	Light soil	Flat	14.1	15.2	45	68	2.0	2.1	282	364	2101	2699	744	744
	6	Light soil	Rolling	13.1	13.9	54	62	5.9	6.1	244	338	1922	2124	298	298
	21	Well drained	Flat	13.0	15.5	33	45	0.7	0.9	267	373	2071	2837	357	680
	22	Well drained	Rolling	10.5	13.6	24	38	1.5	1.9	198	345	1333	2263	185	550
	13	Poorly drained	Flat	13.3	13.7	24	32	1.4	1.5	281	400	1930	2115	427	427
	14	Poorly drained	Rolling	10.3	12.1	24	28	6.2	6.4	253	339	1891	2279	389	389
Wet	7	Light soil	Flat	13.6	15.3	98	141	4.9	5.1	267	355	2064	2826	342	750
	8	Light soil	Rolling	13.5	13.6	58	73	1.8	2	296	413	2142	2452	459	459
	15	Poorly drained	Flat	11.2	12.8	24	43	2.1	2.3	214	286	1699	2539	317	468
	16	Poorly drained	Rolling	13.0	14.7	53	75	3.7	3.9	243	343	1766	2586	177	577
	23	Well drained	Flat	10.0	13.4	48	74	0.8	0.9	208	310	1838	2764	324	575
	24	Well drained	Rolling	13.5	15.0	33	59	1.6	1.8	269	363	1780	2664	189	666
Irrigated	1	Light soil	Flat	17.5	21.5	29	39	0.6	0.8	381	495	2430	3350	264	666
	9	Poorly drained	Flat	17.7	17.7	41	48	1.1	1.3	322	440	2712	3080	622	622
	17	Well drained	Flat	17.3	17.6	97	107	1.7	1.8	340	419	3249	3498	744	744
	3 & 19	Light/Well drained	Rolling	17.3	17.5	88	102	4.0	4.1	352	484	3065	3405	764	764

Table 4 Accumulated analysis of variance showing the relative importance of the main typology attributes and farm intensity on base farm losses of N, P and GHGs.

	Typology attribute and farm intensity	Degrees of Freedom	Sum of Squares[1]	F pr.
Nitrogen loss	Soil drainage	3	6	<0.001
	Soil wetness	3	4	<0.001
	Farm intensity	4	1	0.310
	Slope	1	<1	0.427
Phosphorus loss	Soil wetness	3	55	<0.001
	Soil drainage	3	36	<0.001
	Slope	1	13	0.004
	Farm intensity	4	12	0.093
Methane	Farm intensity	4	130891	<0.001
	Soil wetness	3	75797	<0.001
	Soil drainage	3	30011	<0.001
	Slope	1	41	0.795
Nitrous Oxide	Farm intensity	4	5588071	<0.001
	Soil wetness	3	3822919	<0.001
	Soil drainage	3	2190550	<0.001
	Slope	1	543004	0.177
CO_2 from Urea	Soil wetness	3	499935	<0.001
	Farm intensity	4	416251	<0.001
	Soil drainage	3	245833	<0.001
	Slope	1	20690	0.002

Note: [1]The sum of squares represents the portion of the total variation in the data set that can be attributed to a particular variable.

produced a co-benefit for reducing GHG emissions, with CH_4 emissions reduced by between -2 to 11% and both N_2O and CO_2 reduced by up to 24% (Figure 2B). Removing N fertiliser altogether reduced N losses by 9 to 40%, and P losses by up to 10%. This option also suggested a GHG co-benefit, with CH_4 emissions reduced by 4 to 17%, N_2O emissions reduced by 15 to 43% and CO_2 reduced by between 84 and 100% (Figure 2C).

Off-paddock cow facility mitigations were effective in lowering N losses to water but less effective at lowering P losses (Figure 3). In most cases, off-paddock cow facilities led to a trade-off in GHG emissions, with CH_4 emissions generally increasing while long-lived GHGs showed variable results. The use of a stand-off pad over late autumn and winter generally resulted in a reduction in N leaching, ranging from -4 to 21% while reductions in P losses ranged from -2% to 7%. For a stand-off pad, CH_4 emissions ranged from an increase of 2% to a 3% reduction, N_2O emissions generally increased by up to 6% with no change to CO_2 emissions (Figure 3A). The use of an uncovered wintering pad resulted in N leaching being reduced by between 2 and 43%, while the reduction in P losses varied, ranging from -4 to 20% (Figure 3B). Similarly, the use of a covered wintering barn lowered N losses by between -2 and 38% while changes to P losses varied by between -4 and 20% (Figure 3C). Methane emissions tended to increase with both the uncovered wintering pad and covered barn, ranging between a 1% reduction to an 8% increase. For long-lived gases, the effect on emissions varied widely for the uncovered wintering pad, ranging from a 7% decrease to an 12% increase for N_2O and a 3% decrease to an 12% increase for CO_2 (Figure 3B). For the covered barn, N_2O generally showed a co-benefit effect, with emissions ranging from a 4% increase to a 14% reduction, while CO_2 emissions again ranged from a 3% decrease to an 12% increase.

Increasing the effluent area to optimise the potassium application from the effluent had a varying effect on N losses to water, with effectiveness ranging from -8% to 9% (Figure 4A). Similarly, P losses changed by -11% to 11%. Greenhouse gas emissions were not affected by this mitigation measure, because there was no change in the amount of feed nor the amount of N fertiliser applied. Not applying N fertiliser to the effluent area lowered N losses to water by between 3 and 20% while P losses declined by between 0 and 5%. This mitigation practice had a co-benefit effect on GHG emissions, with CH_4 emissions reduced by up to 4% while N_2O and CO_2 emissions declined by up 17%

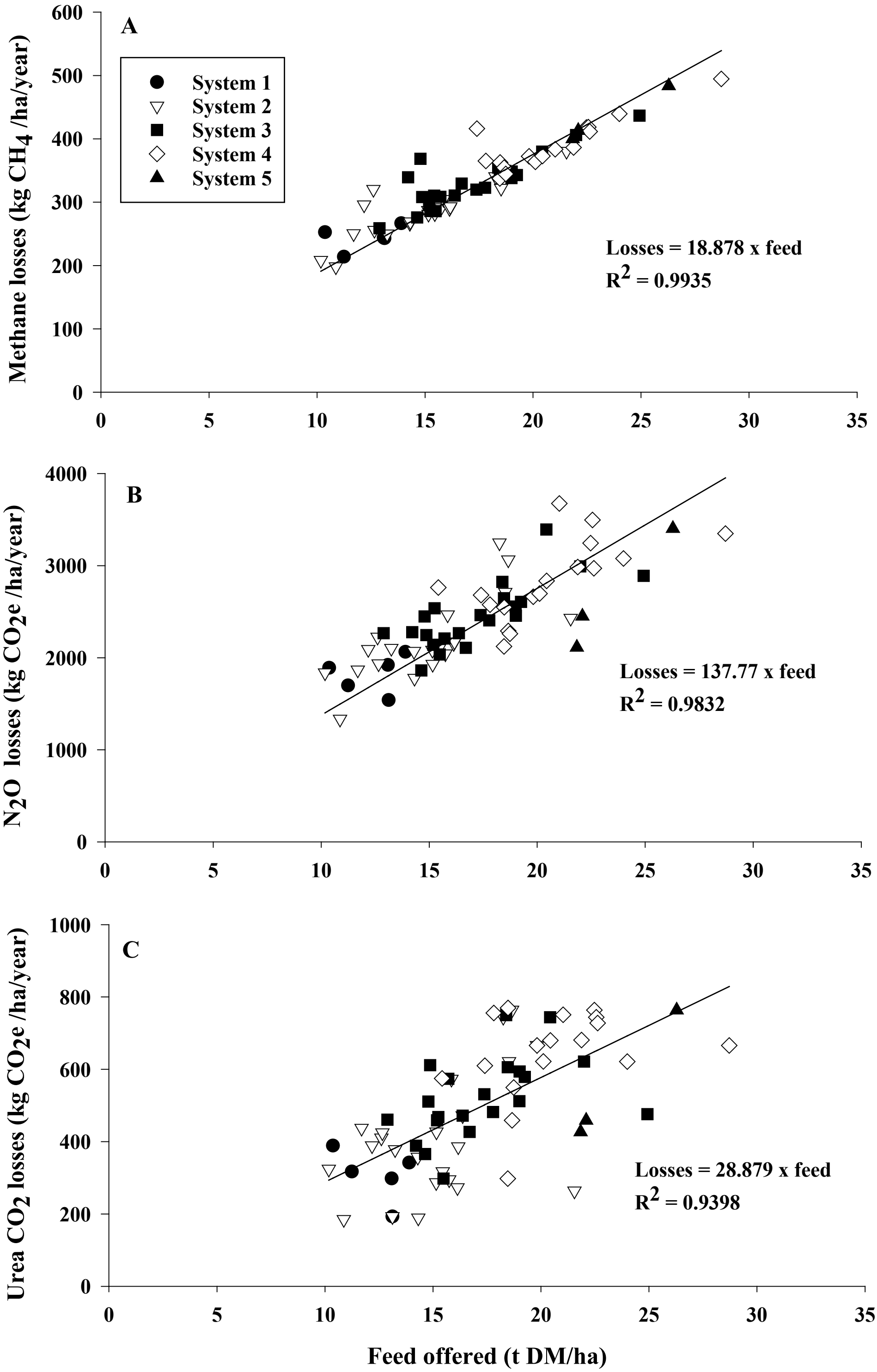

Figure 1 Relationships between total feed offered (kg DM/ha) to animals and losses of A) methane (kg CH_4/ha), B) nitrous oxide (kg CO_{2e}/ha) and C) carbon dioxide from urea fertiliser (kg CO_{2e}/ha). Farm intensity of each base farm indicated by the DairyNZ farm production system definitions (DairyNZ 2021).

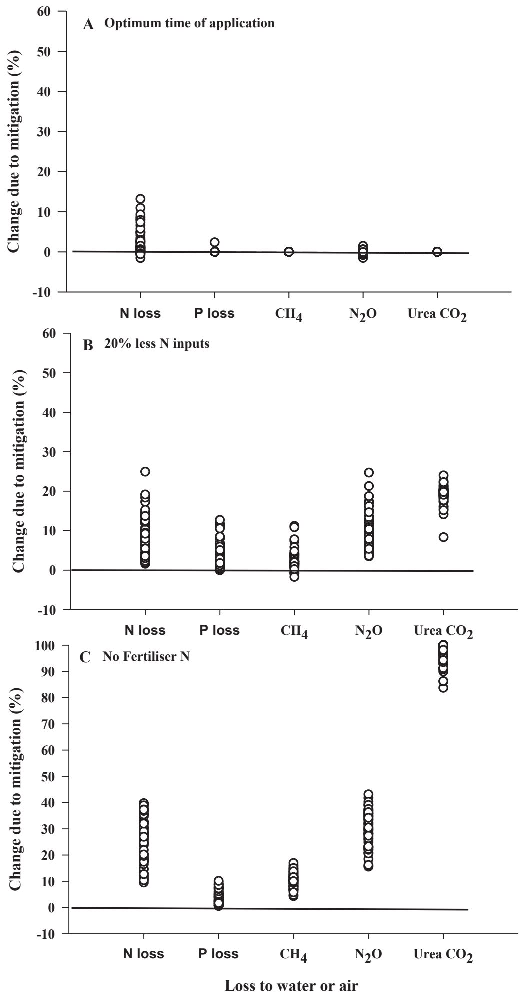

Figure 2 Percentage changes in losses to water and air due to mitigations aimed an **N managemen**t across all relevant dairy typologies, where positive values suggest a reduction in loss and negative values suggest an increase in loss. A) Changes due to optimum months of N application; B) Changes due to lowering N inputs by 20%; C) Changes due to removing N fertiliser. ▨ represents GHG co-benefits (emissions are reduced).

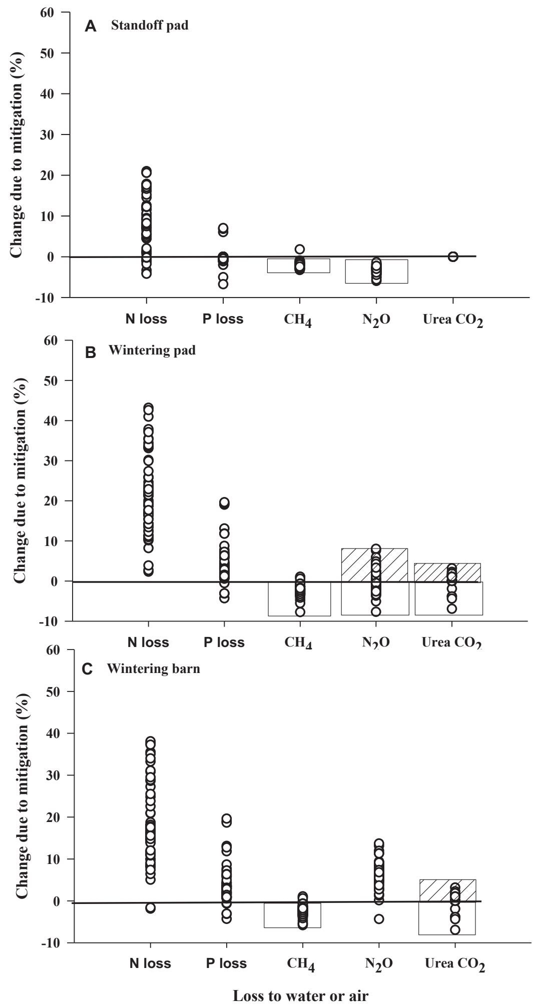

Figure 3 Percentage changes in losses to water and air due to mitigations aimed at **off-paddock cow facilities** across all relevant dairy typologies, where positive values suggest a reduction in loss and negative values suggest an increase in loss. A) Changes due to use of a standoff pad; B) Changes due to use of an uncovered wintering pad; C) Changes due to use of a covered wintering barn. ▨ represents GHG co-benefits (emissions are reduced); □ represents GHG trade-offs (emissions increase)

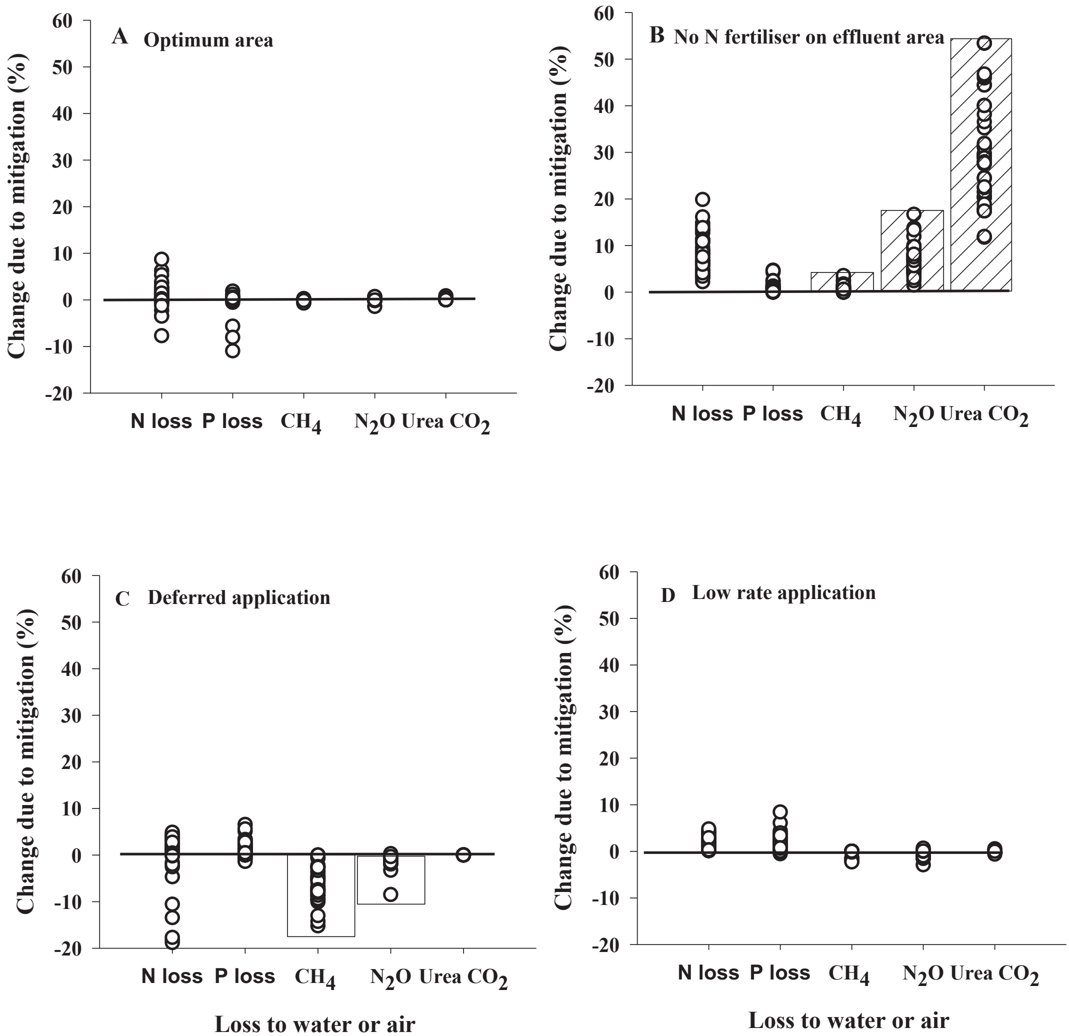

Figure 4 Percentage changes in losses to water and air due to mitigations aimed at effluent management across all relevant dairy typologies, where positive values suggest a reduction in loss and negative values suggest an increase in loss. A) Changes due to increasing effluent area to an optimum size; B) Changes due to applying no N fertiliser to the effluent area; C) Changes due to deferred effluent application; D) Changes due to low-rate effluent application. ▨ represents GHG co-benefits (emissions are reduced); ☐ represents GHG trade-offs (emissions increase).

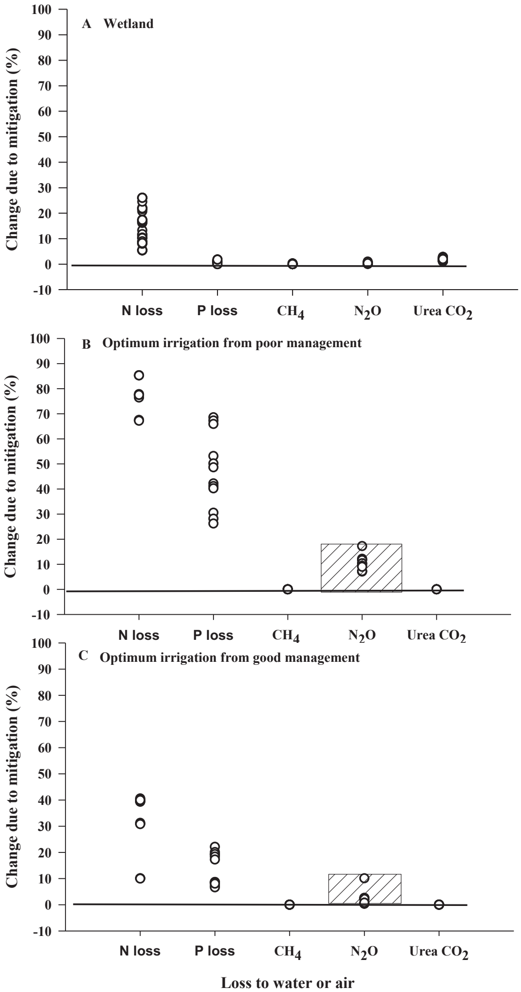

Figure 5 Percentage changes in losses to water and air due to mitigations aimed at edge of field and irrigation management across all relevant dairy typologies, where positive values suggest a reduction in loss and negative values suggest an increase in loss. A) Changes due to adding a wetland; B) Changes due to optimum irrigation management coming from a poor management system (where irrigation was applied at a fixed rate and depth); C) Changes due to optimum irrigation management coming from an improved good management system where a water balance model was used to schedule irrigation. ▨ represents GHG co-benefits (emissions are reduced).

and 54%, respectively (Figure 4B). Deferring effluent application had a widely variable effect on N losses to water, ranging from a 4% reduction in losses to a 19% increase in losses. In contrast, P losses were generally reduced, ranging from -1% to 7% (Figure 4C). The increase in N loss to water was primarily due to the effluent being applied over a shorter period, resulting in higher application rates at times when drainage was still occurring. Deferred effluent application had a trade-off effect on GHG emissions, with CH_4 emissions increasing by up to 15% and N_2O emissions increasing by up to 9%. There were only minimal effects of low-rate effluent application on any of the losses to water or GHG emissions (Figure 4D).

The addition of a wetland was applicable to 14 of the 22 typologies and was effective in lowering N losses to water by between 5 and 26%. However, it had no effect on P losses or GHG emissions (Figure 5A). The irrigation mitigations were also very effective in lowering both N and P losses to water, with the degree of effectiveness depending on the initial management. Where initial management of irrigation application was poor (e.g. based on fixed timing of irrigation and depth of application), implementation of optimum irrigation management (deficit irrigation utilising soil sensors) reduced N losses to water by between 67 and 85% (Figure 5B). Similarly, the P losses could be reduced by between 26 and 69%. Where the farm was already employing a water balance model (WBM) to schedule irrigation, implementation of optimum irrigation management (as noted above, ensuring a soil moisture deficit is maintained) reduced N losses to water by between 10 and 41% while P losses were reduced by 7% to 22% (Figure 5C). Improving irrigation management had no effect on CH_4 emissions but it did reduce N_2O emissions by up to 17%, thus demonstrating a co-benefit.

Discussion

Co-benefits and trade-offs

Co-benefits for GHG mitigation were identified for four of the 12 water quality mitigations tested (Table 5). These were primarily N mitigations, through which a reduction in N input to improve water quality had a consequential reduction in long-lived GHGs (i.e. N_2O), and for some, CH_4. Trade-offs were detected for two water quality mitigations: the use of a stand-off pad and deferring effluent application increased CH_4 and long-lived gases. For the remaining six of 12 water quality mitigations evaluated, either no impact was detected, or a mixture of both trade-offs and co-benefits occurred.

1) Nitrogen management

Judicious scheduling of N fertiliser to avoid high risk months (i.e. winter) has no effect on GHG emissions. While N leaching was slightly reduced, there was very little effect on N_2O emissions for two reasons. Firstly, any reduction in indirect N_2O emissions from leached N was minor because of the small reduction in N leaching. Secondly, while N_2O emissions may be expected to decline by avoiding N fertiliser application to wetter soils when pasture N uptake is limited, modelling of

Table 5 Classifications of water quality (WQ) mitigation measures based on their potential to reduce greenhouse gas emissions. If type of gas is not shown, then no or minimal effect was observed.

Water quality mitigation measure	Impact on GHG emissions	Type of GHG affected by WQ mitigation
Lowering N inputs by 20% (N fertiliser, low N feeds) (note, due to reduced pasture production)	Co-benefits	CH_4, N_2O, CO_2
Removing N fertiliser (note, due to reduced pasture production)		CH_4, N_2O, CO_2
Avoiding N fertiliser on effluent areas (note, due to reduced pasture production)		CH_4, N_2O, CO_2
Improved irrigation management (relative to 'good' and 'poor' management)		N_2O
Avoiding N fertiliser in high-risk months	No or minimal impact	-
Optimum effluent area		-
Change to low-rate effluent application		-
Constructed wetland		-
Wintering pad (June, July)	Mixture of co-benefits and trade-offs	CH_4 (trade-off only); N_2O, CO_2 (trade-off and co-benefit)
Wintering barn (June, July)		CH_4 (trade-off only); N_2O (co-benefit only); CO_2 (trade-off and co-benefit)
Stand-off pad (In use May to August)	Trade-offs	CH_4, N_2O
Deferred effluent application		CH_4, N_2O

N_2O emissions by *Overseer* is currently unable to capture the influence of wet soils. Instead, the losses are estimated based on the national GHG inventory approach of applying an annual average emission factor (Ministry for Primary Industries, 2022). Methane and urea CO_2 emissions did not change as this practice had a minimal effect on, respectively, the feed supply and total amount of urea fertiliser applied.

A 20% reduction in N inputs as synthetic fertiliser will reduce pasture production, resulting in fewer animals and decreased amounts of N excretion (Clark et al. 2020). Furthermore, for the farms where PKE was used, replacing this moderate N supplement (2.7% N content; Wheeler and Watkins, 2015) with a low N supplement will reduce N intake per kg DM. Fewer animals reduced enteric CH_4 and excreta-derived N_2O emissions, while substituting PKE with options containing lower N concentrations reduced urinary N excretion (Selbie et al. 2015). This will reduce N_2O emissions from grazed pastures, given urine patches are the major source of N_2O emissions from dairy farms (de Klein et al. 2020). There was a statistically significant positive relationship between initial total N inputs (as fertiliser and supplements) and the reduction in N_2O emissions across the modelled farms ($R^2 = 0.30$, $P < 0.001$, data not shown). This suggests that the GHG co-benefit from this mitigation measure is relatively larger on farms with high N inputs compared to farms with low N inputs.

Avoiding N fertiliser use reduced pasture production by 5-27% (data not shown), which required a reduction in the number of cows supported on the farms by between 5 and 17%. In turn, less feed consumed will reduce both CH_4 and long-lived GHG emissions, providing a co-benefit.

2) Off-paddock management

Removing cows from wet paddocks onto standoff pads to reduce N leaching (Clark et al. 2020) will invariably lead to an increase in the volume of managed manure. Solids will accumulate on pads while a proportion of the liquid fraction may drain into effluent ponds, thereby increasing the amount of stored manure leading to enhanced GHG emissions. These manure-derived CH_4 and N_2O emissions are greater than from the equivalent base farm with no standoff pad, resulting in a trade-off with respect to GHG emissions. Effluent ponds are the main source of manure-derived CH_4 emissions on NZ dairy farms (Ministry for the Environment, 2023) and any increase in volume will lead to greater emissions. Inclusion of carbon-rich bedding material (sawdust, bark and woodchips) with excreta on off-paddock facilities such as standoff pads can lead to elevated N_2O emissions through increased nitrification and denitrification activity (Groenestein and van Faassen, 1996; van der Weerden et al. 2014). A similar trade-off effect with standoff pads was noted by van der Weerden et al. (2018), who reported a 10% increase in manure/excreta-related GHG emissions (N_2O and CH_4, converted to CO_2 equivalents) per cow when compared to a farm system with no standoff pad. The extent of use of a standoff pad will influence the degree of a trade-off effect.

Uncovered wintering pads with a carbon-rich bedding material resulted in a trade-off with respect to CH_4 emissions, most likely driven by the same processes as for a standoff pad where increased manure storage increases CH_4 emissions. In contrast, the impact on N_2O and CO_2 emissions varied, with both co-benefits and trade-offs being observed (ranging from a 8% reduction to a 12% increase in emissions). The reason for a co-benefit or trade-off outcome appears to be partly influenced by specific typology attributes and partly by farm management practices. A trade-off in N_2O emissions can be expected, as noted for the stand-off pads i.e. potential increase in nitrification and denitrification activity from carbon-rich bedding material leading to increased N_2O emissions. This increase was greater than the N_2O emissions avoided by taking cows off paddocks over the winter months, with the largest increase in emissions occurring on drier, irrigated typologies. Under these conditions, N_2O production from winter soils was apparently lower compared to rain-affected wintering pad manure that was then stored for 5 months prior to field application. However, in most cases we observed a co-benefit, due to reduced N_2O emissions from paddocks being greater than the increase in N_2O emissions from manure collected and stored. This co-benefit was generally greatest in wetter climates with poorly drained soils, from which N_2O emissions can be significant. The trade-off and co-benefits in CO_2 emissions were due to the small variations in urea N fertiliser use.

Wintering barns lead to both a trade-off (increased CH_4 emissions) and co-benefit (reduced N_2O emissions), while CO_2 emissions exhibited both trade-offs and co-benefits**.** Methane emissions increased by up to 6%, due to increased effluent pond emissions. However, a wintering barn with a regularly-scraped concrete floor will produce a slurry-like manure, which when stored, will have negligible N_2O emissions due to the slurry's anaerobic conditions resulting in a high proportion of the available N being completely denitrified to dinitrogen (N_2) (Laubach et al. 2015). These results contrast with an earlier study of an Otago dairy farm system that included a covered barn with woodchip bedding material and a weeping wall storage facility, which indicated an increase in manure/excreta-related GHG emissions (N_2O and CH_4) by 35% per cow compared to a farm with no barn (van der Weerden et al. 2018). The type of barn and manure management system, be it slurry- or organic bedding-based, can have a marked effect on the subsequent GHG footprint (Anestis, unpublished data), directly influencing a potential co-benefit or trade-off. As for the wintering pad, the trade-off and co-benefits in CO_2 emissions were due to the small variations in urea N fertiliser use.

3) Effluent management

Optimising the size of the effluent area for potassium application generally had no effect on GHG emissions, most likely due to this mitigation measure having no influence on the amount of feed grown nor the amount of N fertiliser used.

Avoiding N fertiliser application onto effluent blocks has a similar co-benefit effect as the earlier mitigation measure of reducing N inputs by 20% (described above). In this instance, avoiding N fertiliser on effluent blocks reduced total pasture production leading to lower N_2O emissions (up to 17% reduction), driven by both the reduced amount of N fertiliser and reduced urine deposition due to fewer cows (1% reduction in cow numbers). The co-benefit extended to CH_4 emissions, albeit this was not as large (up to 4% reduction) compared to the decrease in N_2O emissions because the impact on the former was linked solely to the reduced pasture production. Reduced CO_2 emissions were due to the reduced urea fertiliser use. The extent of the GHG co-benefit did not appear to be influenced by any specific typology attributes.

Deferring effluent application to avoid winter months (Monaghan et al. 2007) requires longer effluent storage durations, which will therefore extend the time available for GHG emissions to occur. Thus, this practice has a trade-off effect on both CH_4 and N_2O emissions. Others have also noted the risk of manure storage duration on GHG emissions, noting reduced storage time can be an effective mitigation option (van der Weerden et al. 2014; Laubach et al. 2015), however this approach would conflict with the aim of reducing N leaching (Laubach et al. 2015). Fertiliser use remained unchanged with this mitigation measure, therefore urea-induced CO_2 emissions did not change compared to the base farm. There were no specific typology attributes directly influencing the degree of trade-off for the deferred irrigation option. Outperformed

Moving to low-rate effluent irrigation had little impact on GHG emissions, with minimal changes to CH_4 and long-lived GHG emissions. While low-rate application can be effective at reducing N and P loss for specific soil types and slopes (Monaghan et al. 2007), we would not expect a co-benefit or trade-off effect with the mitigation measure as there is no change to the amount of feed on offer to grazing livestock, nor any impact on the N load.

4) Edge of field attenuation

Wetlands are effective at reducing N leaching but results from the Overseer modelling would suggest these features have little effect on CH_4 and long-lived GHG emissions. A recent literature review concluded N_2O emissions from these constructed features are likely to be very low, to the extent of suggesting the potential for trade-offs is unlikely (Simon et al. 2021). Indeed, first principles would suggest there may be a possibility for co-benefits, due to an overall reduction in N_2O emissions from reduced N leaching and thus reduced indirect emissions from water bodies. However, further research is required to confirm or refute this possible co-benefit. While CH_4 emissions appear to be unchanged, past New Zealand research would indicate lower CH_4 emissions downstream of constructed wetlands relative to upstream, possibly due to the oxygen release from plants within constructed wetlands reducing CH_4 emissions that may otherwise occur (Tanner et al. 1997). There is a need for further research to improve our understanding of the impact of constructed wetlands on GHG emissions, which could then aid further improvements to farm-scale decision support tools such as Overseer.

5) Improved irrigation management

Improved irrigation management (i.e. application of ‘deficit’ irrigation) had a co-benefit with respect to N_2O emissions, but no impact on CH_4 and CO_2 emissions. When irrigation was improved following ‘poor’ management, the co-benefit was greater (average of 10% reduction in N_2O emissions) compared to improvements following ‘good’ management (average reduction of 2% in N_2O emissions). In both scenarios, the reduction in N_2O emissions reflect the reduced N leaching with improved irrigation management (where the reduction was greater for the former scenario), leading to reduced indirect N_2O emissions. Maintaining soil moisture deficits through improved irrigation management may also help to reduce direct N_2O emissions due to soils being below a critical moisture content that favours denitrification. Nitrous oxide emissions increase with increasing soil moisture content, up to about 80% water filled pore space, above which the low soil oxygen concentrations lead to complete denitrification, with available N predominantly lost as N_2 (Balaine et al. 2016). Emissions below a critical moisture content can also be influenced by the rate of return and depth of application of the irrigation (Vogeler et al. 2019).

The modelling data generally support our initial hypotheses, with most N management strategies and both of the improved irrigation management strategies indicating co-benefits in reduced GHG emissions. We hypothesised that improved effluent management may have a varied impact on GHG emissions due to the range of measures; this was borne out in our findings, with most measures having no impact while one (deferred effluent application) indicated a trade-off with increased GHG emissions. Edge of field attenuation via constructed wetlands also showed no impact on GHG emissions, whereas we had hypothesised N_2O emissions would be reduced. Lastly, inclusion of off-paddock cow facilities generally led to a trade-off in GHG emissions, supporting our initial hypothesis, however there were examples where co-benefits (reduced N_2O and CO_2 emissions) occur, most likely through associated changes in fertiliser practices.

Anomalies in mitigation effectiveness on N and P losses

The focus of this paper is on trade-offs and mitigations for GHGs. However, we were also able to use this opportunity to quantify the effectiveness of these mitigations on reducing N and P losses to water. While our results generally aligned closely with Monaghan et al. (2021 a, b,) and McDowell et al. (2020), there were a few cases where the applied mitigations had minimal effect or even increased N and P losses to water. These are worthy of additional discussion.

Off paddock mitigations, in particular the standoff pads, increased N and P losses for several typologies. For N, the increases were likely a result of the timing of solid manure application related to the timing of irrigation (both typologies with higher N losses were irrigated). For those typologies with increased P loss to water, the increased volume of effluent generated by the off-paddock cow facilities, in particular the standoff pad, resulted in both an increased loading of P to the existing area receiving liquid effluent and an increased area receiving solid manure application. The use of standoff pads in high moisture conditions has been shown to increase the risk of P loss due to these very factors (Laurenson et al. 2012). Increasing the effluent area to minimise potassium (K) inputs typically resulted in a reduction in N leaching, but for some typologies an increase in N leaching was observed (Figure 4A). It was a similar story for P loss, where some typologies saw a decrease in P loss while others saw an increase in P leaching. While increasing the effluent area resulted in a lower N loading from the effluent per ha, spreading it over a larger portion of the farm did in some cases increase total farm losses. Additionally, restricting the amount of supplement fed on the effluent area (to limit the K input from supplements) resulted in a higher supplement N loading to the non-effluent area of the farm, again increasing total farm losses in some cases. Not applying effluent over the late autumn-winter period resulted in increased N loadings in effluent over the remaining nine months. For typologies where a considerable amount of drainage occurred in spring and autumn, increased effluent N loadings resulted in increased N loss over those periods.

Conclusion

Four of the 12 mitigations for reducing N and P losses to water had a co-benefit effect on GHG emissions. Improved N management strategies generally showed a co-benefit due to reduced farm N inputs leading to lower pasture production. Improved irrigation management strategies aided in reducing indirect N_2O emissions via reduced N leaching. Constructed wetlands and improved effluent management practices that did not impact N fertiliser use had no effect on GHG emissions, as the amount of feed produced did not alter. However, two of the mitigations (stand-off pads and deferring effluent application) led to increased GHG emissions across most typologies, suggesting a cautious approach is required when considering the suitability and impact of these mitigations at the farm-scale. In both cases, the trade-offs relate to the increased volume and/or duration of stored manure leading to increased CH_4 emissions.

Although we used hypothetical farms that may not necessarily fully represent all farm systems across a wide range of New Zealand dairy typologies, the insights gained provide a first step towards providing pastoral farmers and rural professionals with quantitative data on co-benefits and trade-offs for reducing GHG emissions when mitigating nutrient losses to water. It is recognised that further work will be required to capture the cost effectiveness of these mitigation options. For now, the information presented here can be useful for initial conversations on options to reduce contaminants losses, from which more informed decisions can be investigated for individual farms. Policy makers may also find value in these findings at a higher level when considering options for land use planning at a regional scale.

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