CSIRO PUBLISHING

SPECIAL ISSUE: IDBC 2022 | RESEARCH PAPER
https://doi.org/10.1071/AN22436

ANIMAL PRODUCTION SCIENCE

Effects of water-quality management mitigations on greenhouse-gas emissions from deer farms

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Handling Editor:
Gordon Dryden

Received: 24 November 2022
Accepted: 15 March 2023
Published: 4 April 2023

Cite this:
Stevens DR *et al.* (2023)
Animal Production Science, **63**(16), 1669–1678.
doi:10.1071/AN22436

ABSTRACT

Context. Red deer farming in New Zealand has increased in intensity, increasing the emissions to water and air. Outdoor wintering systems pose a significant threat to water quality through sediment loss and nitrate leaching. Changing wintering systems to bring animals indoors shifts emphasis to greenhouse-gas emissions. **Aims.** To investigate the relative potential emissions to water and air when red deer are wintered outdoors on forage crops or indoors on supplements. **Methods.** The impacts of wintering red deer on forage or indoors were calculated for five farms, involving 32 herds containing 2167 deer over 2 years, in southern New Zealand. Animal classes included weaners, hinds and stags. Potential losses to water included sediment, nitrogen and phosphorous, while losses to air included methane, nitrous oxide and ammonia. Losses to air were calculated using current New Zealand greenhouse-gas inventory calculations and revised calculations recognising published forage, soil and bedding emissions factors not yet included in the inventory. **Key results.** Calculated outdoor winter feed intake was 9.5% greater than indoor measured feed intake. The average herd size of 115 deer wintered indoors for an average of 87 days would have needed 1.8 ha of winter forage crop. Potential losses of sediment, nitrogen and phosphorus were calculated to be 5362, 106 and 5.2 kg per herd respectively, if wintered on crop. Total greenhouse-gas (GHG) emissions calculated using current inventory emission factors were higher if deer grazed a forage crop than when wintered indoors (2.58 vs 2.41 kg CO_2-e/head/day respectively). When revised emission factors were used, indoor wintering produced greater GHG emissions than did wintering outdoors (2.61 vs 2.28 kg CO_2-e/head/day respectively). **Implications.** Variability may occur both in contaminant loss to water and emissions to air. Trade-offs between the two need to be recognised in decision-making. As the science of GHG develops, the relative ranking of different systems may change.

Keywords: air, bedding, feed intake, forage type, greenhouse gas, red deer, water, wintering systems.

Introduction

Environmental degradation remains a significant concern when the utilisation of grasslands intensifies (Merten and Minella 2013). New Zealand grazing systems, evolved over 150 years, have captured significant gains in productivity (Fennessey *et al.* 2016), although are now showing signs of stress (Chapman *et al.* 2022). Red deer farming, initiated in the 1970s, has followed this trend as significant markets for venison and velvet antler have been developed (Gray 2021). In recognition of these challenges to the environment, regulatory frameworks to control emissions to water and air are being implemented in New Zealand (McFarlane *et al.* 2020).

Outdoor wintering systems dominate management practices throughout New Zealand. Forage cropping, using brassicas and fodder beet, provides significant amounts of the feed requirements of livestock, especially in the South Island of New Zealand. Typically, between 3% and 7% of farmland is dedicated to winter forage crops on intensive grazing farms (Stevens *et al.* 2021). However, wintering on forage crops can have a significant impact on nutrient loss (Smith and Monaghan 2020) and soil damage, resulting in sediment loss to waterways (Monaghan et al. 2017). Farmers are responding quickly, exploring options that minimise this impact.

Red deer have a range of behavioural characteristics that exacerbate potential for sediment loss and bacterial contamination of water ways. Fence-line pacing is known to decrease pasture cover and increase erosion risk (McDowell et al. 2004). Wallowing in water, a natural behaviour, also increases potential phosphorus and bacterial loadings in waterways (McDowell and Stevens 2006). Various managements have been proposed to mitigate these activities (McDowell and Stevens 2006), leaving winter management of forage crops as a significant on-going source of contaminants when deer farming (McDowell and Stevens 2008), similar to other outdoor ruminant grazing systems in New Zealand (McDowell et al. 2003).

Wintering systems are a significant source of contaminants to waterways because stock graze at a time of year when drainage or surface runoff occurs. Both runoff and subsurface flows transport contaminants off land and into nearby waterways (Monaghan et al. 2017). Deer grazing both pasture and forage crops during winter contribute to contaminant losses (McDowell and Stevens 2008). While the loss of sediment from forage crops (1010 kg/ha) was approximately 30 times greater than losses from pasture (31 kg/ha), the total amount lost per farm will depend on the percentage of pasture or forage cropping on any farm. Potential losses ranged from 20 to 4480 kg/ha (McDowell and Stevens 2008), indicating the opportunity for management to control contaminant loss.

Avoiding winter forage crops by bringing livestock indoors is one approach to minimising contaminant losses to waterways. Bringing livestock indoors for winter changes the farm dynamic. Grazing practices are replaced by diet and animal health management, while land use is replaced by capital investment in infrastructure. It also has the potential to alter greenhouse-gas (GHG) emissions as home-grown feeds are substituted for imported feed, adding extra processing and transport, and altering emission factors. For example, the digestion of *in situ* kale averaged 28% lower methane emissions (Thomson et al. 2016) than did digestion of grass-based diets for sheep. However, nitrous oxide emissions from urine deposited on soil after forage crop grazing by dairy cows are approximately double those from an equivalent pasture grazing event (derived from van der Weerden et al. (2017)).

Farmers must carefully consider the overall impacts of practice change when faced with the need to reduce both contaminant loss to water and emissions loss to air. In this research, we document the potential to prevent contaminant losses to water while comparing different models of GHG emissions for a group of farmers who were aiming to reduce contamination of water by replacing *in situ* forage crop grazing with winter-housing systems.

Materials and methods

Farm descriptions

Five farmers in the southern region of New Zealand developed indoor facilities (barn wintering) to house herds of deer during winter. These facilities were developed to replace *in situ* grazed winter forage crops as a source of winter nutrition. This allowed the housing of animals to prevent damage to water-logged and erosion-prone soils from winter forage grazing.

Farmers housed a range of livestock classes, most predominantly velveting stags of a range of ages, but including hinds and weaners. In total, 32 herds were used over a 2-year period. The average herd size was 115 animals, with an average indoor period of 87 days (Table 1). Bedding used was either a bark chip or sawdust deep-litter system, with occasional additions during the winter if deemed necessary. A top-soiled layer (approximately 10–20 cm in depth) was removed at the end of each season and refreshed with new material. Bedding lasts between two and four winters. Animals were housed to meet the animal-welfare guidelines specific for deer (Ministry for Primary Industries 2018).

Calculating feed intake and faecal and urinary outputs

Actual feed intake of the animals housed indoors was calculated from data of feedstock used, including silage, baleage and imported concentrate supplements. Nutritive value of the feed provided (Table 2) was determined by near-infrared spectroscopy (NIRS; Hill Laboratories, Hamilton New Zealand).

Feed requirements if each herd was grazed on a forage crop were calculated from equations published by Nicol and Brookes (2007) for New Zealand red deer. These feed requirements were then used, in conjunction with herd numbers and nutritive value of forage crops, to predict the

Table 1. Farm descriptions, including the type of livestock wintered indoors, herd size and length of winter feeding.

Farm	Livestock class[A]	Herd size[B]	Winter length (days)[B]
1	1, 2, 3	55, 55, 70	94, 86, 82
2	3, 4, 5	260, 122, 89	77, 105, 97
3	1,3	40, 210	98, 86
4	5	200	92
5	1,3	184,102	90, 76
Mean		115	87

[A]Stock classes: 1, stags 2 years of age; 2, stags 3–4 years of age; 3, stags >4 years of age; 4, mixed-age hinds; 5, 6–9-month-old mixed-sex weaners.
[B]Herd size and winter-length values in each row correspond to livestock classes in the same row, in the same order.

Table 2. Mean dry-matter content, metabolisable energy concentration and N concentration in feeds supplied to red deer fed indoors or outdoors on a kale forage crop during winter.

Item	Indoor		Outdoor
	Home-grown feed	Imported supplement	Forage crop
Dry matter (%)	26.0	65.4	15.0
Metabolisable energy (MJ/kg)	10.2	10.3	11.4
Nitrogen content (%)	2.35	2.28	2.43

land area required for each herd. A yield of 14 t DM/ha was assumed, on the basis of yields in the region of study published in the literature (Thompson *et al.* 2010; Smith *et al.* 2012; Monaghan *et al.* 2017).

Values of 11.4 MJ ME/kg DM and 152 g CP/kg DM (Westwood *et al.* 2014) were used in calculations to determine dry-matter intake and potential N output. Nitrogen outputs via urine and faeces were determined by the following process:

(a) Nitrogen intake was calculated (dry-matter intake (DMI; kg/day) × N concentration in the feed (g/kg DM))
(b) Faecal output was calculated (intake × (1 − (ME/GE/0.81))) using eqns 1.5 and 1.10 (SCARM 1994)
(c) Nitrogen output in faeces was calculated (−4.623 + (197 × feed N (g N/g DMI)) + (feed intake × 7.89)) × 0.001 (eqn 5.7 from Ministry for Primary Industries (2022))
(d) Nitrogen output in urine was calculated by difference (N intake – N in faeces)

Calculating estimates of contaminant loss

Estimates of losses of sediment, phosphorus (P) and nitrogen (N) were made to indicate the potential reduction in losses by using barn wintering. Monaghan *et al.* (2017) compared two forage crop grazing management approaches over 2 years in a paired-catchment study on soil types that were similar to those farmed by the study group. They concluded that the average loss of sediment, P and N (in both overland and subsurface flow) was 4130, 4.4 and 39 kg/ha.annum respectively, under a standard grazing regime with dairy cows. Using a strategic grazing approach, which avoided critical source-area grazing until the end of the grazing period, average loss of sediment, P and N (in both overland and subsurface flow) was 830, 1.4 and 21 kg/ha.annum respectively. Given that farmers are progressing towards implementing the strategic grazing approach, the averages of these two sets of values (2980, 2.9 and 30 kg/ha.annum for sediment, P and N respectively) have been used to estimate losses. When grazing forage crops *in situ*, some wastage is measured. A feed-utilisation value of 85% (Judson and Edwards 2008) was applied to crop yield to determine total area required for grazing, and hence contributing to potential soil loss.

Loss of nitrogen to air as ammonia (NH_4) also occurs from dung and urine deposition. The New Zealand IPCC inventory methodology (Ministry for Primary Industries 2022) has estimated NH_3 emissions to then calculate indirect N_2O emissions. The NZIPCC methodology value of a 10% loss of N to NH_3 is used to estimate losses to air from grazing winter forage crops. Nitrogen deposited as dung and urine to bedding when wintering indoors also creates ammonia. Measurements reported when dairy effluent was mixed with sawdust (van der Weerden *et al.* 2014) showed that a total of 1.14% and 3.0% of N deposited was lost as N_2O and NH_3 respectively, after 7 months. These values were used to estimate losses to air in this analysis.

Calculating GHG emissions

Calculations of GHG emissions included carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), from production of feed and its transport and from livestock emissions. Emissions from the production and transport of supplementary feed used emission factors published by Ledgard and Falconer (2015) for New Zealand cropping practices (Table 3). These include emissions relating to both fuel use and land-based emissions from cultivation and fertiliser use. Transport emissions were added to reflect delivery of supplementary feed sourced off-farm, at the rate of 100 g CO_2/t.km, assuming an average 100 km round trip from supplier to the farm.

New Zealand runs tier-two country-specific emissions factors for livestock. Methodology from the New Zealand national GHG inventory calculations was used to calculate GHG emissions (Ministry for Primary Industries 2022). Briefly, this allocates GHG emissions on the basis of enteric methane production from digestion, methane from dung degradation, and N_2O from dung and urine deposition to land, using specific factors of N loading in dung and urine (van der Weerden *et al.* 2020). Standard values and equations specific for deer (Ministry for Primary Industries 2022) were used to calculate emissions from digestion (methane), dung deposition (methane and N_2O) and urine deposition (N_2O) (Table 4). Adjustments were made to these calculations to

Table 3. Carbon footprint of supplementary feed and forage crops in New Zealand (from Ledgard and Falconer 2015).

Feed type	kg CO_2-equivalent/kg DM
Palm kernel expeller	0.506
Grain concentrate	0.355
Pasture silage	0.201
Hay	0.182
Kale	0.192
Cereal silage	0.185

Table 4. Values for methane production and nitrous oxide loss used to calculate greenhouse-gas emissions.

Item	IPCC nomenclature	NZ IPCC Inventory[A]	Wintering type	
			Barn	Forage crop
Enteric methane (g/kg DM intake)	$CH_{4\ enteric}$	21.25	21.25	15.3[B]
Dung methane (g/kg DM deposited)	$CH_{4\ PRP}$	0.915	0.915	0.915
N_2O emission factor for dung (kg N_2O-N/kg N)	$EF_{3PRP\ DUNG}$	0.0012	na	0.0012
N_2O emission factor for urine (kg N_2O-N/kg N)	EF_{3PRP}	0.0074	na	0.0148[C]
N_2O emission factor for barn manure (kg N_2O-N/kg N)	$EF_{3(S\ SS)}$	na	0.0114[D]	na
Fraction of N from urine lost through NH_3 volatilisation (kg NH_3-N/kg N)	$Frac_{GASM\ URINE}$	0.10	0.03[D]	0.10
N_2O emission factor for NH_3 volatilisation (kg N_2O-N/kg N)	EF_4	0.01	0.01	0.01

[A]Values from the New Zealand greenhouse-gas inventory methodology (Ministry for Primary Industries 2022).
[B]Forage crop value reduced by 28% as per Thomson *et al.* (2016).
[C]Value = 2 × inventory value; derived from literature data in Table 4.
[D]From van der Weerden *et al.* (2014).

reflect variations due to forage crop digestion and soil conditions under winter forage crop grazing.

A meta-analysis of measurement of methane emissions from the enteric digestion of forage crops (Thomson *et al.* 2016) reported changes in methane emissions from +13% to −64%. The average change in methane emissions was −28% from both sheep and cattle. The methane emissions (g CH_4/kg DM consumed) determined by the standard calculations were reduced by this amount (Table 4) for deer fed forage crops.

Nitrous oxide emissions from soil increases when saturated and disturbed by forage crop grazing (van der Weerden and Styles 2012; Monaghan *et al.* 2013; Di *et al.* 2016; Treweek *et al.* 2016; van der Weerden *et al.* 2017). To reflect the greater emissions from disturbed exposed soils post-forage grazing, the loss of N deposited as urine was adjusted as per results documented in Table 4. The average N to N_2O loss of N deposited to forage crop soil was of 1.94% from dairy cows. This is two times that emitted from urine of cattle deposited on pastures. Using this logic, the emissions from deer urine deposited to soils during forage crop grazing was doubled from 0.74% N lost as N_2O for pasture to 1.48% N lost as N_2O for forage crops (Table 5).

These were converted to CO_2 equivalents using current Global Warming Potential (GWP) factors, as per animal emissions (Ministry for Primary Industries 2022).

$$CO_2 \text{ equivalents} = CH_4 \times 27.2 + N_2O \times 273.$$

Data were summarised to compare standard inventory calculations with calculations customised for the wintering barn or forage crop situations. Contaminant results are presented at the herd level for wintering method (barn or forage crop) and stock type (rising 2-year-old stags, rising 4-year-old stags, mixed-age stags, mixed-age hinds, mixed-sex weaners). GHG emissions are presented at the individual animal level.

Table 5. Literature values for nitrous oxide emission factors from dairy cattle grazing winter forage crops.

Source	Crop grazed	N lost as N_2O (% of N deposited as dung and urine)
van der Weerden *et al.* (2017)	Kale	0.75
Monaghan *et al.* (2013)	Kale	1.41
Treweek *et al.* (2016)	Kale	2.1
Di *et al.* (2016)	Kale	1.1
van der Weerden and Styles (2012)	Swedes	3
van der Weerden and Styles (2012)	Swedes	3.3
Mean response		1.94
N loss from pastures grazed with dairy cows		0.98

Results

Estimated intake

Estimates of individual intake of red deer in barns, from records of feed stockpile use, increased with animal size (Table 6). Calculated potential forage crop intakes were often numerically greater than the barn recorded intake, averaging 0.26 kg DM/day or 9.5% more than barn intakes. This difference was greatest for mixed-age stags at 0.95 kg DM/day. Nitrogen intakes varied at the same level of magnitude as dry-matter intake, although were 20% greater on the forage crop, due to higher dry-matter intake (Table 6) and higher N content (Table 2).

Estimates of contaminant losses

Estimates of potential contaminant loss when herds graze a forage crop were proportional to the area required for grazing (Table 7). Therefore, mixed-age stags, with both the greatest calculated intake and the largest average herd size, incurred

Table 6. Estimates of feed and nitrogen intake in barn-fed and forage-fed red deer of a range of genders and ages.

Livestock class	Total number of animals wintered	Winter feeding period (day)	Intake (kg DM/head.day)		Nitrogen intake (kg N/head.day)	
			Barn	Forage crop	Barn	Forage crop
Rising 2-year-old stags	536	90	2.68	2.95	0.061	0.072
Rising 4-year-old stags	110	86	3.75	3.60	0.080	0.087
Mixed-age stags	860	80	3.30	4.25	0.073	0.103
Mixed-age hinds	217	98	2.04	2.19	0.045	0.053
Mixed-sex weaners	444	83	1.83	1.93	0.043	0.047
Mean	433	87	2.72	2.98	0.060	0.072

Barn estimates were made from farmer records of amount fed, while forage estimates were calculated (Nicol and Brookes 2007).

Table 7. Estimates of grazing area required and losses of P, nitrate, ammonia and sediment from soils under forage crops grazed by red deer during winter.

Livestock class	Average herd size (*n*)	Area grazed (ha)	P loss (kg/herd)	Nitrate loss (kg/herd)	Ammonia loss (kg/herd)	Sediment loss (kg/herd)
Rising 2-year-old stags	77	1.91	5.5	57.2	55.1	5675
Rising 4-year-old stags	55	1.43	4.1	42.9	41.3	4259
Mixed-age stags	123	3.86	11.2	115.9	111.7	11 505
Mixed-age hinds	36	1.10	3.2	33.0	31.8	3272
Mixed-sex weaners	49	0.71	2.0	21.1	20.4	2099
Mean	68	1.80	5.2	54.0	52.0	5362

the greatest estimated losses. Nitrate losses and ammonia losses were approximately equal and were both an order of magnitude greater than was P loss (Table 7).

Methane emissions

The calculation of methane emissions from enteric fermentation and from dung (Table 8) demonstrated the differences that may occur as emission factors, intake and digestibility vary. For example, methane emissions from dung were influenced by digestibility of the diet, as faecal output was reduced on the forage crop, resulting in lower emissions from dung (Table 8). Methane from enteric digestion was calculated to be approximately 10% greater when red deer were wintered on a forage crop when inventory values were used, due to a higher intake. However, when the revised value was used, deer fed forage crops had calculated methane emissions that averaged 21% less than those for the deer fed indoors.

Nitrous oxide emissions

Calculated nitrous oxide emissions from dung were approximately 10% greater from the forage crop than were the barn inventory calculations (Table 9). However, when revised to reflect greater potential loss from the interaction with the sawdust bedding, the calculated direct N_2O emissions from dung were an order of magnitude greater for the barn system (Table 9). The revised barn calculation was 65% more than the inventory calculation for urine N_2O losses (Table 9). The revised forage crop calculation was 100% greater than the inventory calculation for urine N_2O losses (Table 9).

Indirect N_2O losses from NH_3 volatilisation were numerically similar when calculated using the inventory calculation. The revised barn calculation resulted in a reduction in indirect N_2O losses from NH_3 volatilisation by approximately 68% (Table 9).

Total N_2O losses using the inventory calculations were 22% greater on the forage crop than in the barn (Table 9). However, the revised calculations estimated that the forage crop emissions were approximately 5% lower than were barn emissions. Both revised barn and forage crop emissions were approximately 100% greater than inventory estimates.

Total GHG-emission estimates

Greenhouse-gas estimates from animal activities by inventory calculation were higher from the forage crop than barn system (Table 10). When revised to reflect changes due to differences in enteric CH_4 emissions from forage crops and N_2O and NH_3 losses, emissions from deer housed in barns were 11% higher than inventory calculations, while those from deer fed on forage crops were approximately 16% lower than inventory calculations.

Table 8. Estimates of methane emissions from barn-fed or forage-fed red deer using standard inventory calculations or revised calculations.

Item	Livestock class	System			
		Barn		Forage crop	
		Inventory	Revised	Inventory	Revised
Methane from dung (g/day)	Rising 2-year-old stags	0.98	0.98	0.91	0.91
	Rising 4-year-old stags	1.07	1.07	0.77	0.77
	Mixed-age stags	0.78	0.78	0.63	0.63
	Mixed-age hinds	0.61	0.61	0.47	0.47
	Mixed-sex weaners	0.53	0.53	0.42	0.42
	Mean	0.80	0.80	0.64	0.64
Methane from digestion (g/day)	Rising 2-year-old stags	70.21	70.21	90.27	65.00
	Rising 4-year-old stags	79.58	79.58	76.43	55.03
	Mixed-age stags	56.99	56.99	62.58	45.06
	Mixed-age hinds	43.30	43.30	46.60	33.55
	Mixed-sex weaners	38.88	38.88	41.01	29.53
	Mean	57.79	57.79	63.38	45.63

Revised calculations included recent research findings of lower enteric emissions from the digestion of brassica crops (Thomson *et al.* 2016).

Greenhouse-gas emissions from the provision of food to the barn or forage crop deer were similar (Table 10). Total GHG emissions followed the same pattern as those produced by animal activities, with the revised calculations being higher than inventory calculations for the barn system (+8%), while being lower for the forage crop system (−12%).

Discussion

Calculated contaminant losses to the environment were considerable when forage crops were fed to deer during the winter. Larger animals such as mixed-age stags with greater daily feed requirements had a larger impact through requiring a greater area for food provision. Various researchers have documented the extent of losses when red deer graze (e.g. de Klein *et al.* 2003; McDowell and Stevens 2006, 2008), ranging from 440 to more than 2200 kg soil/ha.annum, with losses from winter crop grazing estimated at approximately 1000 kg soil/ha.annum (McDowell and Stevens 2008). The estimated values in the current study were greater than those reported in the literature. This may be related to several factors that influence potential contaminant loss. These include the type of animal used, the duration of grazing/soil disturbance and the soil type itself.

The use of dairy cattle data to provide the estimates may be influenced by the size of the animal, and the damage done by the downward pressure applied to the soil. Soil disturbance by the hooves of the grazing animal is a major contributor to potential soil loss and is much greater with cattle than sheep (Donovan and Monaghan 2021). For example, the downward pressure applied by a 670 kg dairy cow with a hoof area of approximately 50 cm^2 has been measured to be 180–200 N/cm^2 at push-off during walking (van der Tol *et al.* 2003). During this action, this relates to the transfer of approximately half of the weight of the animal to the area of hoof still in contact with the ground. van der Tol *et al.* (2003) measured this area to be approximately one-third of the resting area of the hoof, or 15 cm^2. When translated into metrics for red deer, a stag has an estimated hoof area of approximately 36 cm^2 and a liveweight of approximately 300 kg. Using the relative data from van der Tol *et al.* (2003), this would translate into a force of approximately 120–130 N/cm^2, just over half that exerted by the dairy cows reported by van der Tol *et al.* (2003). The liveweight of the red deer in this study typically sits between dairy cows and sheep, leading to the potential to result in lower soil losses when grazing forage crops than the dairy cow estimates that were used here.

Duration of grazing, or the length of time the animals walk over the soil, also affects potential contaminant loss, as the number of hoofprints increases. The data reported by McDowell and Stevens (2008) from deer represented a 2-week grazing period, while Monaghan *et al.* (2017) reported on an 8-week period of dairy cattle grazing, potentially increasing soil damage by four-fold.

Soil type, too, can have a significant impact on soil loss under intensive winter grazing due to relative susceptibility to physical degradation (Hewitt and Shepherd 1997). The Pallic soil grazed by dairy cattle (Monaghan *et al.* 2017) are known to be structurally vulnerable (Hewitt and Shepherd 1997) and have been recorded to have a resistance strength of approximately 0.2–0.3 MPa when saturated during winter crop grazing (Thompson *et al.* 2010), which is approximately 10% of the downward force of the dairy cow reported by van der Tol *et al.* (2003). This confirms the vulnerability of

Table 9. Estimates of nitrous oxide emissions from barn-fed or forage-fed red deer using standard inventory calculations or revised calculations.

Item	Livestock class	System			
		Barn		Forage crop	
		Inventory	Revised	Inventory	Revised
N_2O from dung (g/day)	Rising 2-year-old stags	0.049	0.572	0.064	0.064
	Rising 4-year-old stags	0.055	0.645	0.054	0.054
	Mixed-age stags	0.040	0.465	0.044	0.044
	Mixed-age hinds	0.030	0.351	0.033	0.033
	Mixed-sex weaners	0.027	0.321	0.029	0.029
	Mean	0.040	0.471	0.045	0.045
N_2O from urine (g/day)	Rising 2-year-old stags	0.545	1.038	0.809	1.618
	Rising 4-year-old stags	0.588	1.121	0.684	1.369
	Mixed-age stags	0.460	0.876	0.560	1.120
	Mixed-age hinds	0.342	0.652	0.417	0.833
	Mixed-sex weaners	0.335	0.556	0.342	0.685
	Mean	0.454	0.848	0.562	1.125
N_2O from NH_3 (g/day)	Rising 2-year-old stags	0.114	0.034	0.162	0.162
	Rising 4-year-old stags	0.125	0.038	0.137	0.137
	Mixed-age stags	0.095	0.029	0.112	0.112
	Mixed-age hinds	0.071	0.021	0.084	0.084
	Mixed-sex weaners	0.068	0.019	0.070	0.070
	Mean	0.095	0.028	0.113	0.113
Total N_2O (g/day)	Rising 2-year-old stags	0.707	1.644	1.034	1.843
	Rising 4-year-old stags	0.768	1.804	0.876	1.560
	Mixed-age stags	0.594	1.369	0.717	1.277
	Mixed-age hinds	0.443	1.024	0.533	0.950
	Mixed-sex weaners	0.430	0.896	0.442	0.784
	Mean	0.589	1.347	0.720	1.283

Revised calculations included recent research findings for emissions from urine deposited on soil during forage crop grazing (van der Weerden and Styles 2012; Monaghan *et al.* 2013; Di *et al.* 2016; Treweek *et al.* 2016; van der Weerden *et al.* 2017) and manure deposition to sawdust (van der Weerden *et al.* 2014).

these soils to damage during the grazing of a winter forage crop, increasing the potential for contaminant loss (Donovan and Monaghan 2021). The soils in the deer study reported by McDowell and Stevens (2008) were, by the description of the site, most likely Brown soils, which are much more resistant to physical damage (Hewitt and Shepherd 1997).

Contaminant loss can be reduced using good grazing management practices to ensure that soil damage is minimised through lower livestock density (offering multiple days of feed allocation at a time), and/or through minimising time spent on each allocation of land (daily shifting to a new forage area for example). These reduce the relative grazing intensity and result in reduced soil damage (Donovan and Monaghan 2021). Measures to prevent grazing of critical source areas also reduce potential soil losses from winter forage crop grazing (Monaghan *et al.* 2017).

Overall, calculated GHG emissions are similar to those in other literature. Most reports are from work conducted for whole years, rather than short periods such as this. However, comparison can be made using the yield of GHG per kg DM eaten. For example, the output of animal GHG emissions from dry-matter intake using the standard calculation method in this study is approximately 0.65 kg CO_2-e/kg DM, similar to the 0.67 kg CO_2-e/kg DM that can be calculated from data reported by Vibart *et al.* (2021) for low-intensity sheep and beef farming.

More intensive dairy farming systems include more supplementary feed and winter forage cropping, increasing the contribution from sources such as cultivation and the mechanical processes associated with this and supplementary feed conservation. The total GHG output calculated for dairy cattle from van der Weerden *et al.* (2018) is 0.8–0.85 kg CO_2-e/kg DM. Inventory calculations provided values of 0.87 and 0.89 kg CO_2-e/kg DM for barn and forage crop systems respectively. The revised calculations of the current data set were 0.96 and 0.77 kg CO_2-e/kg DM, for barn and

Table 10. Estimates of total greenhouse-gas emissions from barn-fed or forage-fed red deer using standard inventory calculations or revised calculations.

Item	Livestock class	System			
		Barn		Forage crop	
		Inventory	Revised	Inventory	Revised
Greenhouse-gas emissions from animal activities (kg CO_2-e/head.day)[A]	Rising 2-year-old stags	1.73	1.94	1.88	1.59
	Rising 4-year-old stags	2.40	2.68	2.32	1.94
	Mixed-age stags	2.13	2.38	2.72	2.30
	Mixed-age hinds	1.32	1.47	1.40	1.19
	Mixed-sex weaners	1.19	1.31	1.23	1.03
	Mean	1.75	1.95	1.91	1.61
Greenhouse-gas emissions from food provision (kg CO_2-e/head.day)[B]	Rising 2-year-old stags	0.63	0.63	0.67	0.67
	Rising 4-year-old stags	0.83	0.83	0.81	0.81
	Mixed-age stags	0.79	0.79	0.96	0.96
	Mixed-age hinds	0.49	0.49	0.50	0.50
	Mixed-sex weaners	0.54	0.54	0.44	0.44
	Mean	0.66	0.66	0.67	0.67
Total greenhouse-gas emissions (kg CO_2-e/head.day)	Rising 2-year-old stags	2.36	2.56	2.55	2.26
	Rising 4-year-old stags	3.24	3.51	3.13	2.76
	Mixed-age stags	2.92	3.16	3.68	3.26
	Mixed-age hinds	1.81	1.96	1.90	1.68
	Mixed-sex weaners	1.73	1.85	1.66	1.46
	Mean	2.41	2.61	2.58	2.28

[A]Includes CH_4 from enteric fermentation and dung decomposition, and N_2O from dung and urine deposition and from NH_3 volatilisation.
[B]Includes CO_2 from cultivation, harvesting, transport and feeding out, plus N_2O emissions from soils during cultivation.

forage crop systems respectively. While these values are higher than those calculated from data reported by van der Weerden *et al.* (2018), the values in the current data set represent only 87 days, or 24% of the year. These 87 days are the most intensive, so the expectation is that transitioning back to a pasture-based feeding regime would result in a decrease in intensity. Assuming the remainder of the year had outputs similar to Vibart *et al.* (2021), an overall GHG output of approximately 0.70 kg CO_2-e/kg DM would be the result.

Both revised calculations of N_2O emissions were nearly double that predicted by the inventory calculations. Soil conditions have a significant role in altering N_2O emissions, especially from urine deposition. Compaction increased N_2O emissions from 0.3 to 0.75 kg N_2O/kg N applied (van der Weerden *et al.* 2017). This may be soil related as the opposite has occurred when soil compaction is very high, although not with urine deposition, and the emissions factor was 3.2 kg N_2O-N/kg N (van der Weerden and Styles 2012). Smith *et al.* (2008) measured an emissions factor of 1.4 N_2O-N/kg N from urine deposited during the grazing of swedes. All may depend on extent of compaction and anaerobic conditions. Prolonged treading damage on saturated soils can cause failure of the soil structure leading to anaerobic conditions in some forage grazing situations (McDowell *et al.* 2003). Emissions from cattle dung deposition average 0.005 kg N_2O/kg N applied (van der Weerden *et al.* 2021). Incorporation into the soil will occur with crop use but did not make any difference to N_2O emissions, leading to the standard application of an emission factor of 0.006 by the IPCC.

Ammonia from cattle manure applied as a slurry is also included in international inventories (van der Weerden *et al.* 2021). Incorporation into the ground (crop) has an emissions factor of 0.129, while the emissions factor for broadcast application is 0.242. If it remains a solid, then the NH_3 emission factor is reduced to 0.03. These large variations in potential emissions may have significance to values calculated on-farm when moving from outdoor to indoor systems. The New Zealand inventory calculations may be lacking some elements to adequately assess the GHG from barn systems as barn manure emission factors are not presently included. Emissions from sawdust-based bedding altered the pathway of emissions (van der Weerden *et al.* 2017).

Methane emissions were reduced when forage type was accounted for. Thomson *et al.* (2016) summarised nine experiments that compared a range of winter forage types, including kale, rape, turnip, swede and fodder beet, with control diets, mainly of ryegrass pasture. A reduction of methane emissions was recorded in 27 of 28 treatments feeding winter forages. These ranged from −2.4% to −61% compared with the control and were related most closely to the dietary NDF concentration (R^2 = 0.40). This provides some confidence that the emission factor applied to winter forages should be lower than that for the mainly pasture-based supplements used in the barn feeding systems.

The data presented here demonstrated some of the variability that may occur both in contaminant loss to water and emissions to air. It also provided an indicator of the potential difference in GHG emissions that may be possible as knowledge of the emission processes are refined.

References

Chapman D, Stevens D, King W, Kerr G, Dodd M, Catto W, Caradus J (2022) Greener pastures: priorities for a low-footprint, high-value food producing future. In 'A supplement to the resilient pastures symposium, grassland research and practice series 17'. (Ed. MJ Casey). (New Zealand Grassland Association: Mosgiel, New Zealand) doi:10.33584/rps.17.2021.3604

De Klein CAM, Drewry JJ, Nagels JW, Scarsbrook M, Collins R, McDowell RW, Muirhead R (2003) Environmental impacts of intensive deer farming in New Zealand: a review. In 'The nutrition and management of deer on grazing systems'. (Ed. MJ Casey) pp. 79–88. (New Zealand Grassland Association: Wellington, New Zealand)

Di HJ, Cameron KC, Podolyan A, Edwards GR, de Klein CAM, Dynes R, Woods R (2016) The potential of using alternative pastures, forage crops and gibberellic acid to mitigate nitrous oxide emissions. *Journal of Soils and Sediments* **16**, 2252–2262. doi:10.1007/s11368-016-1442-1

Donovan M, Monaghan R (2021) Impacts of grazing on ground cover, soil physical properties and soil loss via surface erosion: a novel geospatial modelling approach. *Journal of Environmental Management* **287**, 112206. doi:10.1016/j.jenvman.2021.112206

Fennessey PF, Glennie SF, McCorkindale AB (2016) Innovations behind the farm gate that will influence performance of hill farming. In 'Hill country symposium. Grassland research and practice series 16'. (Ed. ER Thom) pp. 15–20. (New Zealand Grassland Association: Dunedin, New Zealand). 10.33584/rps.16.2016.3253

Gray L (2021) 'In Hindsight: 50 years of deer farming in New Zealand.' (Quentin Wilson Publishing: Christchurch, New Zealand)

Hewitt AE, Shepherd TG (1997) Structural vulnerability of New Zealand soils. *Australian Journal of Soil Research* **35**, 461–474. doi:10.1071/S96074

Judson HG, Edwards GR (2008) Survey of management practices of dairy cows grazing kale in Canterbury. *Proceedings of the New Zealand Grassland Association* **70**, 249–254. doi:10.33584/jnzg.2008.70.2704

Ledgard S, Falconer S (2015) Total greenhouse gas emissions from farm systems with increasing use of supplementary feeds across different regions of New Zealand. Final report to MPI. Ministry for Primary Industries, Wellington, New Zealand. p. 70.

McDowell RW, Stevens DR (2006) Experiments examining soil and water quality in a New Zealand deer farm. In 'Focus on environmental research.' (Ed. EB Davis) pp. 235–257. (Nova Science Publishers)

McDowell RW, Stevens DR (2008) Potential waterway contamination associated with wintering deer on pastures and forage crops. *New Zealand Journal of Agricultural Research* **51**(3), 287–290. doi:10.1080/00288230809510459

McDowell RW, Drewry JJ, Muirhead RW, Paton RJ (2003) Cattle treading and phosphorus and sediment loss in overland flow from grazed cropland. *Australian Journal of Soil Research* **41**, 1521–1532. doi:10.1071/SR03042

McDowell RW, Drewry JJ, Paton RJ (2004) Effects of deer grazing and fence-line pacing on water and soil quality. *Soil Use and Management* **20**, 302–307. doi:10.1111/j.1475-2743.2004.tb00374.x

McFarlane K, Sinner J, Campos C, Clapcott J (2020) Environmental limits – a proposed framework for Aotearoa New Zealand. Report to the Ministry for the Environment. Cawthron Institute Report number 3530. p. 187. Available at https://environment.govt.nz/assets/Publications/Files/environmental-limits.pdf [Accessed 1 May 2022]

Merten GH, Minella JPG (2013) The expansion of Brazilian agriculture: soil erosion scenarios. *International Soil and Water Conservation Research* **1**(3), 37–48. doi:10.1016/S2095-6339(15)30029-0

Ministry for Primary Industries (2018) Deer: code of welfare. p. 51. Available at https://www.mpi.govt.nz/dmsdocument/46027/direct [Accessed 1 May 2022]

Ministry for Primary Industries (2022) Methodology for calculation of New Zealand's agricultural greenhouse gas emissions (8). Available at http://www.mpi.govt.nz/news-and-resources/publications/ [Accessed 1 May 2022]

Monaghan RM, Smith LC, de Klein CAM (2013) The effectiveness of the nitrification inhibitor dicyandiamide (DCD) in reducing nitrate leaching and nitrous oxide emissions from a grazed winter forage crop in southern New Zealand. *Agriculture, Ecosystems & Environment* **175**, 29–38. doi:10.1016/j.agee.2013.04.019

Monaghan RM, Laurenson S, Dalley DE, Orchiston TS (2017) Grazing strategies for reducing contaminant losses to water from forage crop fields grazed by cattle during winter, *New Zealand Journal of Agricultural Research* **60**(3), 333–348. doi:10.1080/00288233.2017.1345763

Nicol AM, Brookes IM (2007) The metabolisable energy requirements of grazing livestock. In 'Pasture and supplements for grazing animals'. (Eds PV Rattray, IM Brookes, AM Nicol) pp. 151–172. (New Zealand Society of Animal Production: Hamilton, New Zealand)

SCARM (1994) 'Feeding standards for Australian livestock: ruminants.' p. 266. (CSIRO: Melbourne, Vic., Australia)

Smith LC, Monaghan RM (2020) Nitrogen leaching losses from fodder beet and kale crops grazed by dairy cows in southern Southland. *Journal of New Zealand Grasslands* **82**, 61–71. doi:10.33584/jnzg.2020.82.444

Smith LC, de Klein CAM, Monaghan RM, Catto WD (2008) The effectiveness of dicyandiamide in reducing nitrous oxide emissions from a cattle-grazed, winter forage crop in Southland, New Zealand. *Australian Journal of Experimental Agriculture* **48**, 160–164. doi:10.1071/EA07262

Smith LC, Orchiston T, Monaghan RM (2012) The effectiveness of the nitrification inhibitor dicyandiamide (DCD) for mitigating nitrogen leaching losses from a winter grazed forage crop on a free draining soil in northern Southland. *Proceedings of the New Zealand Grassland Association* **74**, 39–44. doi:10.33584/jnzg.2012.74.2882

Stevens DR, Canton M, Harpham S (2021) Farming with reduced winter cropping in southern New Zealand: the risks and practicalities. *Journal of New Zealand Grasslands* **83**, 123–134. doi:10.33584/jnzg.2021.83.3500

Thompson BR, Stevens DR, Knowles I (2010) Investigating the potential of kale to remove soil potassium accumulated from farm dairy effluent application. *Proceedings of the New Zealand Grassland Association* **72**, 263–267. doi:10.33584/jnzg.2010.72.2793

Thomson BC, Hammond KJ, Muir PD (2016) A review of greenhouse gas emissions from the use of brassica and fodder beet forages on New Zealand farms. A report to the New Zealand Agricultural Greenhouse Gas Centre, Palmerston North, New Zealand. p. 39. Available at https://www.nzagrc.org.nz/publications/a-review-of-greenhouse-gas-emissions-from-the-use-of-brassica-and-fodder-beet-forages-on-new-zealand-farms/ [Accessed 1 May 2022]

Treweek G, Di HJ, Cameron KC, Podolyan A (2016) Effectiveness of the nitrification inhibitor dicyandiamide and biochar to reduce nitrous oxide emissions. *New Zealand Journal of Agricultural Research* **59**, 165–173. doi:10.1080/00288233.2016.1161651

van der Tol PPJ, Metz JHM, Noordhuizen-Stassen EN, Back W, Braam CR, Weijs WA (2003) The vertical ground reaction force and the pressure distribution on the claws of dairy cows while walking on a flat substrate. *Journal of Dairy Science* **86**(9), 2875–2883. doi:10.3168/jds.S0022-0302(03)73884-3

van der Weerden TJ, Styles TM (2012) Reducing nitrous oxide emissions from grazed winter forage crops. *Proceedings of the New Zealand Grassland Association* **74**, 57–62. doi:10.33584/jnzg.2012.74.2884

van der Weerden TJ, Luo J, Dexter M (2014) Addition of straw or sawdust to mitigate greenhouse gas emissions from slurry produced by housed cattle: a field incubation study. *Journal of Environmental Quality* **43**, 1345–1355. doi:10.2134/jeq2013.11.0452

van der Weerden TJ, Styles TM, Rutherford AJ, de Klein CAM, Dynes R (2017) Nitrous oxide emissions from cattle urine deposited onto soil supporting a winter forage kale crop. *New Zealand Journal of Agricultural Research* **60**(2), 119–130. doi:10.1080/00288233.2016.1273838

van der Weerden T, Beukes P, de Klein C, Hutchinson K, Farrell L, Stormink T, Romera A, Dalley D, Monaghan R, Chapman D, Macdonald K, Dynes R (2018) The effects of system changes in grazed dairy farmlet trials on greenhouse gas emissions. *Animals* **8**, 234. doi:10.3390/ani8120234

van der Weerden TJ, Noble AN, Luo J, de Klein CAM, Saggar S, Giltrap D, Gibbs J, Rys G (2020) Meta-analysis of New Zealand's nitrous oxide emission factors for ruminant excreta supports disaggregation based on excreta form, livestock type and slope class. *Science of The Total Environment* **732**, 139235. doi:10.1016/j.scitotenv.2020.139235

van der Weerden TJ, Noble A, de Klein CAM, Hutchings N, Thorman RE, Alfaro MA, Amon B, Beltran I, Grace P, Hassouna M, Krol DJ, Leytem AB, Salazar F, Velthof GL (2021) Ammonia and nitrous oxide emission factors for excreta deposited by livestock and land-applied manure. *Journal of Environmental Quality* **50**, 1005–1023. doi:10.1002/jeq2.20259

Vibart R, Rennie G, Hutchinson K, Burtt A, Chrystal J, Dynes R (2021) Greenhouse gas emissions from New Zealand sheep and beef farms. *Journal of New Zealand Grasslands* **83**, 225–232. doi:10.33584/jnzg.2021.83.3501

Westwood CT, Cutts MK, Russell RJ, O'Brien KM (2014) Effect of timing of harvest on nutritive value of four cultivars of kale (*Brassica oleracea* L. var. *acephala*). *Proceedings of the New Zealand Grassland Association* **76**, 135–140. doi:10.33584/jnzg.2014.76.2970

Data availability. The data that support this study will be shared upon reasonable request to the corresponding author.

Conflicts of interest. The authors declare no conflicts of interest.

Declaration of funding. Deer Industry New Zealand and AgResearch provided funding for this research.

Acknowledgements. The authors thank Deer Industry New Zealand and AgResearch for funding this research. The authors thank the farmers who provided the data.

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