

Scotland's Rural College

Mitigating nitrous oxide and manure-derived methane emissions by removing cows in response to wet soil conditions

van der Weerden, TJ; Laurenson, S; Vogeler, I; Beukes, PC; Thomas, SM; Rees, RM; Topp, CFE; Lanigan, G; de Klein, CAM

Published in:
Agricultural Systems

DOI:
[10.1016/j.agry.2017.06.010](https://doi.org/10.1016/j.agry.2017.06.010)

Print publication: 01/09/2017

Document Version
Peer reviewed version

[Link to publication](#)

Citation for published version (APA):

van der Weerden, T.J., Laurenson, S., Vogeler, I., Beukes, P.C., Thomas, S.M., Rees, R.M., Topp, C.F.E., Lanigan, G., & de Klein, C.A.M. (2017). Mitigating nitrous oxide and manure-derived methane emissions by removing cows in response to wet soil conditions. *Agricultural Systems*, 156, 126 - 138.
<https://doi.org/10.1016/j.agry.2017.06.010>

General rights

Copyright and moral rights for the publications made accessible in the public portal are retained by the authors and/or other copyright owners and it is a condition of accessing publications that users recognise and abide by the legal requirements associated with these rights.

- Users may download and print one copy of any publication from the public portal for the purpose of private study or research.
- You may not further distribute the material or use it for any profit-making activity or commercial gain
- You may freely distribute the URL identifying the publication in the public portal ?

Take down policy

If you believe that this document breaches copyright please contact us providing details, and we will remove access to the work immediately and investigate your claim.

1 **Title changed to: Mitigating nitrous oxide and manure-derived methane emissions by**
2 **removing cows in response to wet soil conditions**

3

4 **Mitigating farm-scale greenhouse gas emissions by removing cows in response to wet**
5 **soil conditions.**

6

7

8 **Authors:** T.J. van der Weerden^{1A}, S. Laurenson², I. Vogeler³, P.C. Beukes⁴, S.M.
9 Thomas⁵, R.M. Rees⁶, C.F.E. Topp⁶, G. Lanigan⁷, C.A.M. de Klein¹

10 **Addresses:**

11 ¹ AgResearch, Invermay Agricultural Centre, Private Bag 50034, Mosgiel, New Zealand.

12 ² AgResearch, Lincoln Research Centre, Private Bag 4749, Christchurch, New Zealand.

13 ³ AgResearch, Ruakura Research Centre, Private Bag 3123, Hamilton, New Zealand.

14 ⁴ DairyNZ, Private Bag 3221, Hamilton, New Zealand.

15 ⁵ Plant and Food Research, Private Bag 4704 Christchurch, New Zealand

16 ⁶ Scotland's Rural College, West Mains Road, Edinburgh EH9 3JG, UK

17 ⁷ Teagasc, Johnstown Castle, Environmental Research Centre, County Wexford, Ireland

18

19 **No. of Tables:** 4

20 **No. of Figures:** 6

21

22 ^A Corresponding author: Tel +64-3-4899012, fax +64-3-4893739

23 Email: tony.vanderweerden@agresearch.co.nz

24 **Abstract**

25 In pasture-based grazing systems, urine deposition is the major source of the greenhouse
26 gas nitrous oxide (N₂O). Livestock treading damage and high soil water contents increase
27 the risk of N₂O emissions. Duration controlled grazing (DCG) practices that are
28 implemented in response to soil water conditions above a threshold may therefore provide
29 an effective means of reducing greenhouse gas (GHG) emissions from dairy farms. In this
30 study we used the DairyNZ Whole Farm Model and APSIM model to assess the cost-
31 benefit of implementing DCG to reduce total N₂O and manure-derived CH₄ emissions
32 from dairy farms. We modelled scenarios on poorly drained or imperfectly drained soils in
33 four regions of New Zealand including Waikato, Manawatu, Canterbury and Southland,
34 where the grazing time on wet days was 0, 13, 17 or 21 hours per day. Emissions were
35 estimated using a refined version of New Zealand's current national greenhouse gas
36 inventory methodology. Our analysis suggested that reducing the grazing time from 21
37 hours to 0, 13 or 17 hours per day when soils were wet could reduce annual N₂O and
38 manure-derived CH₄ emissions by up to, respectively, 12, 9 or 5% on farms with poorly
39 drained soils. The 13 hour per day grazing duration was the least costly, particularly if

40 there were more than 150 ‘wet’ days per year. In contrast, for dairy farms on imperfectly-
41 drained soils, DCG increased emissions, suggesting this management approach for
42 reducing GHG emissions is not suitable for these soils.

43 **Keywords:** Modelling, Whole Farm Model, APSIM, nitrous oxide, duration controlled
44 grazing.

45 1 Introduction

46 Nitrous oxide (N₂O) is an important anthropogenic greenhouse gas (GHG), with
47 agriculture its largest source (Reay et al., 2012). About one third of these global emissions
48 are attributed to excreta returns during livestock grazing (Oenema et al., 1997). Grazing
49 livestock excrete 75-90% of their nitrogen (N) intake in concentrated urine and dung
50 patches (Whitehead, 1995). When deposited on land, the urine-N returns, ranging from
51 200 to 2000 kg N ha⁻¹ for cattle (Selbie et al., 2015), exceed plant uptake capacity and can
52 lead to significant N losses through leaching (Ryden et al., 1984) and gaseous N emissions,
53 including N₂O and ammonia (NH₃) (de Klein et al., 2001). Both N leaching and NH₃
54 emissions are sources of indirect N₂O emissions (Butterbach-Bahl and Dannenmann,
55 2011). In New Zealand, ruminant livestock excreta deposition onto pastures is the single
56 largest source of N₂O, contributing c. 80% of the direct and indirect N₂O emissions (de
57 Klein et al., 2006). Under urine patches, N₂O production and emission will be primarily
58 influenced by oxygen availability which is regulated by soil water content (Linn and
59 Doran, 1984; de Klein et al., 2006). N₂O emission factors have been developed for dairy
60 urine deposited on pasture that incorporate soil water content (van der Weerden et al.,
61 2014). A lower oxygen diffusion rate in soils that have been compacted as a result of
62 animal treading damage can further promote N₂O emissions via denitrification (Ball et al.,
63 2012, van Groenigen et al., 2005).

64 The New Zealand dairy industry aims to increase milk production and reduce greenhouse
65 gas emissions, and acknowledges the challenge in achieving these, sometimes, opposing
66 objectives (Beukes et al., 2011). One particular farm practice that may achieve both
67 objectives is duration controlled grazing (DCG) during wet periods of the year, whereby
68 cow grazing times are reduced with time spent on off-paddock facilities (e.g. standoff
69 pads) for a part of the day. The reduction in grazing hours reduces the amount of excreta N
70 deposited onto wet soils, thereby reducing direct and indirect (via NO₃ leaching) N₂O
71 emissions (de Klein et al., 2006; Christensen et al., 2012; Luo et al., 2013). This practice
72 also protects soils from animal treading damage (Houlbrooke et al., 2009), which in turn
73 may lead to increased pasture production, and, through careful pasture management, can be
74 converted into increased milk production. Measurements reported by de Klein et al. (2006)
75 from southern New Zealand showed that DCG reduced N₂O emissions and NO₃ leaching
76 from paddocks by approximately 40% when cows were on pasture for 3 hours per day
77 during March, April and May compared to 21 hours (normal rotational grazing practices,
78 allowing 3 hours for milking per day). Similarly, in northern New Zealand, Luo et al.
79 (2013) observed 55% reduction in N₂O emissions during spring (September and October)
80 when cow grazing hours during winter (June to August) were reduced from 24 to 6 hours
81 per day.

82 Adoption of DCG practices will increase the volume of excreta that is captured and stored
83 from the off-paddock facility (Luo et al., 2013). Any increase in the volume of excreta
84 stored in manure management systems will increase N₂O, NH₃ and methane (CH₄)
85 emissions from this component of the farm system (Chadwick et al., 2011; Laubach et al.,
86 2015). Therefore, there is potential that DCG practices may lead to ‘pollution swapping’,
87 whereby the emissions from increased manure management potentially over-ride
88 corresponding reductions achieved from avoiding grazing of wet paddocks. Furthermore,
89 the period of time cows are removed from the paddock invariably increases operational

90 costs such as those associated with supplying a quality feed supplement, effluent
91 management and maintenance of the stand-off facilities.

92 A recent analysis of GHG mitigation options showed that a calendar-based approach (i.e.
93 removing cows every day over a certain timeframe e.g. spring) to using standoff pads was
94 not cost-effective (Adler et al., 2015). In order to meet both economic/production and
95 environmental (avoiding pollution swapping) objectives, it is important that cows are
96 removed from paddocks only when it is necessary to do so.

97 Ultimately, farmers will be attracted to options that provide on-farm production and/or
98 financial benefits. Therefore, the objective of this study was to investigate whether tactical
99 removal of dairy cattle from wet paddocks could provide a cost-effective option for
100 reducing farm-scale N₂O and manure-derived CH₄ emissions. To achieve this objective,
101 we (i) developed a relationship between soil volumetric water content (VWC) and N₂O
102 emissions from urine deposition, (ii) modelled excreta cycling and N losses for typical
103 dairy farms in the Waikato, Manawatu, Canterbury and Southland regions of New Zealand,
104 (iii) employed a refined version of New Zealand's greenhouse gas inventory methodology
105 based on the latest available science, and (iv) assessed the cost:benefit of this approach for
106 reducing greenhouse gas emissions. This final step was achieved by utilising the modelled
107 productivity and economic results of implementing DCG when soils were wet, reported in
108 an associated paper (Laurenson et al., submitted).

109 **2 Methodology**

110 2.1 Overview of approach

111 We used a combination of models and existing knowledge to assess the impact of DCG
112 scenarios on N₂O and manure-derived CH₄ emissions for case study dairy farms in four
113 regions of New Zealand: Waikato, Manawatu, Canterbury and Southland. For each farm
114 we used the DairyNZ Whole Farm Model (WFM; Beukes et al., 2008) to estimate excreta
115 N deposition for a 'baseline' farm and three scenarios that included varying grazing
116 durations on days when soils were wet (see section 2.2). Modelled excreta N for each farm
117 scenario was used to estimate direct N₂O emissions employing N₂O emission factors based
118 on a relationship between soil VWC and N₂O emissions (section 2.3). The urine N
119 excretion values estimated by the WFM were also used within the Agricultural Production
120 Systems Simulator (APSIM; Holzworth et al., 2014) modelling framework to assess N
121 leaching and NH₃ emissions from urine patches and N fertiliser for the different farms and
122 scenarios under three rainfall regimes (section 2.4). Leaching losses from dung deposited
123 in the paddock and manure (solid or liquid) from the off-paddock facility were estimated
124 using WFM modelled N loading rates combined with the N leaching fraction used in the
125 New Zealand N₂O inventory methodology (section 2.5). Manure-derived CH₄ emissions
126 were estimated using a combination of the New Zealand IPCC inventory methodology and
127 the default IPCC approach (IPCC, 2006; Ministry for the Environment, 2015). For
128 comparative purposes we also estimated farm-scale N₂O and manure-derived CH₄
129 emissions using emission factors from the NZ GHG inventory methodology (section 2.6).
130 The cost:benefit of the proposed DCG approach was estimated using modelled farm
131 operating profits (Laurenson et al., submitted) and estimated GHG emissions, and is
132 expressed as \$/kg carbon dioxide equivalents (CO₂e) reduction achieved through the
133 adoption of DCG (section 2.7).

134 2.2 Modelling excreta N deposition

135 The DairyNZ WFM was used for estimating excreta N production. This model has been
136 used in New Zealand to model farm management strategies and productivity for a range of
137 pastoral dairy systems (Beukes et al., 2008). A full description of the WFM model can be
138 found in Beukes et al. (2013). In brief, the model framework represents a pasture-based
139 dairy farm with individual paddocks and cows simulated on a daily time step. Cow feed

140 intake is driven by metabolic demand determined by a mechanistic and dynamic model
141 within the WFM that simulates critical elements of cow digestion and metabolism
142 (Hanigan et al., 2009). The cow model predicts daily milksolids production (MS = fat +
143 protein), outputs of N in urine, faeces and milk N output, and methane emissions. The
144 pasture-soil model in WFM (Romera et al., 2009) is climate-driven using daily weather
145 data accessed from the National Institute of Water and Atmospheric Research Virtual
146 Climate Station (VCS) network (Tait et al., 2006).

147 We determined excreta N deposition by modelling dairy farms in four regions including
148 Waikato, Manawatu, Canterbury and Southland that were located on either poorly drained
149 or imperfectly drained soils (Table 1). We used the same soil characteristics for poorly
150 drained and imperfectly drained soils within each region to allow a comparison of the
151 impact of contrasting regional climates on the effectiveness of DCG to reduce GHG
152 emissions. It is important to note that individual simulated farms did not include
153 combinations of both soil drainage classes. The poorly drained soil, a Temuka clay loam, is
154 classified as a Typic Orthic Gley soil by the New Zealand soil classification (Hewitt, 2010;
155 47% clay in top 100 mm) or Mollic Endoaquept by USDA soil taxonomy (Soil Survey
156 Staff, 1998). The imperfectly-drained soil, a Hatfield silt loam, is classified as a Typic
157 Immature Pallic (Hewitt, 2010; 20% clay in top 100 mm) or Udic Haplustept (USDA soil
158 taxonomy; Soil Survey Staff, 1998). Cow stocking rate (SR) was set at a level which
159 ensured that the simulated farms were suitably stocked relative to the pasture grown (Table
160 1). All regions used the same SR for the poorly and imperfectly drained soils, apart from
161 Southland, where the SR for the poorly drained soil was slightly higher (3.15) than for the
162 imperfectly-drained soil (2.75) due to the large difference in typical pasture production
163 across soils in this region (Laurenson et al., submitted).

164

Insert Table 1

165 Duration controlled grazing was imposed when a field's soil VWC exceeded a critical
166 water content (CWC) at the time of grazing. This CWC was defined as the VWC when the
167 risk of treading damage is at its greatest (Piwowarczyk et al., 2011), and varied with soil
168 drainage class. Cows were removed from paddocks if the VWC was greater than 85% of
169 field capacity (FC) on poorly drained soils and 105% of FC on imperfectly drained soils
170 (Laurenson et al., in prep). We compared the CWC with the modelled soil water balance to
171 estimate how many days per year cows should be removed from paddocks due to a risk of
172 treading damage. On the days when VWC > CWC, grazing time per day was either 0
173 hours (i.e. complete removal), 13, 17 hours or 21 hours, where 21 hours represented the
174 baseline in which no restriction was placed on grazing duration. The 0, 13 and 17 hours
175 related to, respectively, 21, 8 or 4 hours on an off-paddock facility (standoff pad). The
176 standoff pad was assumed to have a pine bark and sawdust base (Luo et al., 2008) and was
177 located within 250 m of the milking parlour. It was assumed that cows remained on pasture
178 year round in warmer northern regions (Waikato and Manawatu) where winter pasture
179 growth meets feed demand. In the cooler southern regions, non-lactating cows were
180 'wintered off' farm between 1 June and 8 August, reflecting typical dairy farm practice.
181 Therefore, this analysis considered 365 days of the year in the two northern regions, while
182 the assessment was restricted to the 270 days lactation season (commencing 9 Aug) in the
183 two southern regions.

184 When DCG was not imposed, animals were either on the paddocks, on a lane or in parlour
185 and yards. The amount of urine-N excreted onto these surfaces was proportional to the
186 time spent on each. Cows spent 1 hour per day on lanes and, during the lactation season, 2
187 hours per day in the dairy parlour and yards, with the remaining time was spent on
188 paddocks. Outputs from the WFM included the amount of N deposited as dung and urine
189 onto paddocks, dairy parlour and yards, lanes and standoff areas; the volume of effluent
190 collected, stored and applied to the soil; production and economics data from each
191 simulation. The latter model output has been reported in an accompanying paper
192 (Laurenson et al., submitted).

193 2.3 Relationship between soil water content and N₂O emissions

194 Previous research has shown that N₂O emission factors (EF₃ which quantifies the
195 percentage of applied N lost as N₂O) for dairy cattle urine are strongly related to the soil
196 water filled pore space (WFPS) averaged over 30 days following urine deposition (van der
197 Weerden et al., 2014). For the current study, we adopted VWC as the soil water metric, as
198 it has the advantage of being relatively easy to determine using field sensors and directly
199 compatible with soil water balances under field conditions (van der Weerden et al., 2012).
200 Using N₂O and soil type data from 31 field trials (collated from de Klein et al., 2003, 2004;
201 Luo et al., 2008; Sherlock et al., 2003a,b; Thomas et al., unpubl. data and van der
202 Weerden et al., 2011) we employed the APSIM model to estimate VWC at various soil
203 depths (75, 150 and 200 mm) and for different number of days following urine deposition
204 (15, 20, 30, 45 and 60 days). We then investigated which depth and number of days
205 produced the strongest relationship between modelled VWC and measured EF₃.

206 2.4 Estimating N leaching and NH₃ emissions from urine and N fertiliser

207 As the WFM does not calculate nitrate (NO₃) leaching and NH₃ emissions, we used the
208 estimated amount of excreta N as input parameters to the APSIM model. In New Zealand
209 APSIM has been validated against a range of drainage and leaching regimes that occur
210 under urine-patch conditions (Cichota et al., 2012; 2013). Pasture growth is simulated
211 using AgPasture (Li et al., 2011), with a ryegrass clover mixture, the SoilN and
212 SurfaceOM modules (Probert et al., 1998) were used to describe the C-N cycle, and
213 SWIM2 for the transport of water and solutes, which is based on the Richards' equation
214 and the convection-dispersion equation and the Micromet module (Snow and Huth, 2004)
215 for computing evapotranspiration and energy partition. Also included was a module
216 accounting for volatilisation from urine patches and N fertiliser based on the approach by
217 Générmont and Cellier (1997).

218 Monthly values of urine patch N load (kg/ha) per day, as obtained from the WFM, were
219 used in the APSIM modelling framework to generate estimates of N leaching and NH₃
220 emissions. APSIM simulations ran for a two year period following urine deposition to
221 ensure that all leached N was accounted for. Within a given paddock, N leached from the
222 urine patch were aggregated with N leached from non-urine affected area thereby
223 providing a single N leaching value (Vogeler et al., 2013). A similar approach was taken
224 for modelling and aggregating NH₃ emissions from urine patches and N fertiliser
225 applications. Annual N fertilisation rates differed between regions and farm scenarios,
226 ranging from 68 to 254 kg N/ha. Fertiliser N rates were reduced to account for any N
227 applied in farm dairy effluent (FDE) collected from the standoff pad and solid manure
228 scraped from the pad. It was assumed 85% and 40% of the total N in FDE and solid
229 manure, respectively, would become available for pasture uptake (Gutser et al., 2005;
230 Webb et al., 2013).

231 2.5 Estimating N leaching and NH₃ emissions from dung, effluent and solid manure

232 Paddock N inputs as dung, solid and liquid manure were estimated using the WFM, with
233 effluent applied as necessary (Laurenson et al. submitted). As APSIM has not been
234 validated for N losses from dung, solid and liquid manure, subsequent N leaching and NH₃
235 emissions were based on the New Zealand N₂O inventory methodology, where it was
236 assumed, respectively, 7% and 10% of N inputs were leached as NO₃ and volatilised as
237 NH₃ (Ministry for the Environment, 2015).

238 2.6 Farm-scale N₂O and manure-derived CH₄ emissions from modelled dairy farms

239 Direct N₂O emissions from paddocks are reported as kg N₂O-N/ha/year, and were
240 calculated using the VWC function (previously described in section 2.3) for determining
241 cattle urine EF₃. Total N₂O and manure-derived CH₄ emissions were calculated for each
242 dairy farm scenario and reported on the basis of kg CO₂e/ha/year, where N₂O and CH₄
243 have global warming potentials of 298 and 25 times that of CO₂, respectively, over a 100-

244 year time horizon, as used by the IPCC (Forster et al., 2007). These total emissions were
245 calculated using a refined version of the New Zealand IPCC inventory methodology
246 (Ministry for the Environment, 2015). Key refinements include (i) N₂O emissions from
247 urine deposited onto paddocks estimated using the relationship developed between VWC
248 and N₂O emission factors, and (ii) improved estimation of NH₃ and NO₃⁻ losses from urine
249 and urea fertiliser using a modelling approach (APSIM); Table 2 lists all refinements and
250 assumptions employed. We also categorised all excreta deposited onto standoff pads as
251 ‘solid storage’, based on the definitions of manure management systems (Table 10.18,
252 IPCC 2006). IPCC default values were employed except for direct N₂O emissions from
253 solid storage (EF_{3s}), where we used results from a New Zealand study (EF_{3s} = 0.01%; Luo
254 and Sagar, 2008). We also assumed 4% of total N excreted onto standoff pads drained
255 into FDE ponds, based on research by Luo et al. (2008). We estimated CH₄ emissions from
256 standoff pads (kg CH₄/cow/year) by assuming volatile solids (VS) were 3.5 kg dry
257 matter/cow/day, corrected for the time on the standoff, maximum CH₄ producing capacity
258 for manure from cattle (B₀) was 0.24 m³ CH₄/kg VS and a CH₄ conversion factor (MCF, %) of
259 4% (equation 10.23, IPCC 2006). Modelling was conducted for three individual years
260 for each region, representing years when rainfall depth was equivalent to the 10th, 50th and
261 90th percentile for years between 1995 and 2014. Presentation and discussion of modelling
262 data focuses primarily on results from the 50th percentile rainfall year (20-year average),
263 while data from all modelled years were used when analysing cross-regional relationships.

264

Insert Table 2

265 We excluded the CH₄ emissions from enteric fermentation from all calculations of total
266 greenhouse gas emissions, as the modelled farms maintained the same annual dry matter
267 intake per cow and therefore the same CH₄ emissions (Clark et al., 2003) regardless of
268 whether cows remained on, or were removed from, paddocks. We also present farm-scale
269 N₂O and manure-derived CH₄ emissions based on the current inventory methodology, as a
270 comparison to the refined approach. Paddock-derived N₂O emissions were estimated using
271 the New Zealand-specific EF₃ value of 1% of urine N deposited, as employed in the
272 current New Zealand N₂O inventory. The current inventory methodology does not account
273 for manure collected on standoff pads. Therefore, it was assumed that all off-paddock
274 excreta deposition would be accounted for as effluent stored in anaerobic lagoons, as is
275 currently conducted within the New Zealand agricultural greenhouse gas inventory.

276 2.7 Cost:benefit of DCG

277 The financial cost or benefit from adopting DCG was calculated from the change in dairy
278 operating profit (Table 3), as determined from the economics component of the WFM
279 (Beukes et al., 2013) and total N₂O and manure-derived CH₄ emissions (current study).
280 The dairy operating profit considered the most relevant farm variables (e.g. sale of MS and
281 culled stock, enterprise costs such as insurance, labour expenses and farm system capital
282 and operating costs). The cost-benefit was based on a long term milksolids (MS) price of
283 NZ\$6 per kg MS, and is presented as \$/kg CO₂e reduction achieved through the adoption
284 of DCG.

285

Insert Table 3

286 **3 Results**

287 3.1 Direct N₂O emissions from urine deposition onto paddocks

288 Nitrous oxide emissions from pastoral soils increased with soil water content due to
289 anaerobic conditions stimulating denitrification activity. The strongest relationship
290 between soil water content and EF₃ was observed when VWC in the top 75 mm of soil was
291 averaged over 20 days following urine deposition (VWC_{20d}; R² = 0.42; P < 0.001; Fig.1).
292 Using this relationship, modelled N₂O emissions from urine deposited onto paddocks
293 ranged from 2.6 to 2.7 kg N₂O-N/ha/year from the poorly drained soils in all four regions

294 when cows remained on paddocks (Fig. 2a, 2c, 2e and 2g). When cows were completely
295 removed from wet paddocks, emissions from poorly drained soils in the two South Island
296 regions were reduced by 38-54%, while a reduction of 76-82% was predicted for farms in
297 the two North Island regions. In contrast, N₂O emissions from the imperfectly-drained soil
298 were low when cows remained on wet soils due to the relatively lower VWC, with
299 emissions ranging from 0.54-0.78 kg N₂O-N/ha/year. Completely removing cows from
300 paddocks when imperfectly drained soils were wet reduced paddock-derived N₂O
301 emissions by 49-59% in the two North Island regions, whereas a relatively small reduction
302 of 6% was calculated for the South Island farms due to cows wintered off in June and July
303 which reduced the frequency of grazing events that occurred on 'wet' days.

304 Emissions of N₂O from urine deposition based on the current IPCC methodology are
305 estimated as the product of N load and EF₃, where the latter has a value of 1%, regardless
306 of soil water content. Therefore, for the baseline, N₂O emissions from urine deposition
307 were the same for the two soil drainage classes within each region in Waikato, Manawatu
308 and Canterbury since the amount of urine-N deposition (i.e. N load) was the same (Fig. 2b,
309 2d and 2f). In contrast, Southland showed slightly higher N₂O emissions per hectare for the
310 poorly drained soil when DCG was not implemented (Fig. 2h) due to slightly higher
311 stocking rate at 3.15 cows/ha compared to 2.75 cows/ha for imperfectly drained soils and
312 therefore N load onto the soil (Table 1). Implementing DCG when soils were wet reduced
313 direct N₂O emissions from paddocks in all regions (Fig. 2b, 2d, 2f and 2h), reflecting the
314 lower amount of urine N that was deposited onto pasture and the lower EF value for
315 standoff pads (0.0001; Table 2).

316 *Insert Figure 1*

317 *Insert Figure 2*

318 3.2 Farm-scale N₂O and manure-derived CH₄ emissions from dairy farms

319 We consider the refined inventory methodology provides a more accurate assessment of
320 the impact of our DCG strategy on total N₂O and manure-derived CH₄ emissions at the
321 farm-scale. However, we include a comparison with the current New Zealand inventory
322 methodology (section 3.3) to illustrate the difference in total N₂O and manure-derived CH₄
323 emission estimates between the two methodologies.

324 *Baseline*

325 When DCG was not implemented, total N₂O and manure-derived CH₄ emissions ranged
326 from 1667 to 2656 kg CO₂e/ha/year for imperfectly drained soils and from 3015 to 3785
327 kg CO₂e/ha/year for poorly drained soils (Fig. 3). Manure-derived CH₄ emissions
328 represented 37-51% and 29-33% of the total N₂O and manure-derived CH₄ emissions for the
329 imperfectly and poorly drained soils, respectively. Direct and indirect N₂O emissions from
330 excreta deposition, fertiliser application and manure storage and application (exclusive of
331 CH₄ emissions from manure management) ranged from 817 to 1457 kg CO₂e/ha/year for
332 imperfectly drained soils, and 2027 to 2552 kg CO₂e/ha/year for poorly drained soils (Fig.
333 3).

334 For imperfectly drained soils in Manawatu and Canterbury, the largest contribution to
335 direct N₂O emissions was from N fertiliser (40% and 35% of total N₂O emissions,
336 respectively). Whereas, for farms on imperfectly drained soils in Waikato and Southland
337 and on poorly drained soils in all regions, urine deposited directly onto pasture was the
338 largest N₂O source accounting for between 32% and 67% of total N₂O emissions. We
339 explored cross-regional relationships by utilising modelling results from all three modelled
340 years (10th, 50th and 90th percentile rainfall years). Using results from the baseline farms
341 (i.e. DCG not implemented) on two contrasting soil drainage classes in four regions, we
342 observed a significant linear relationship between the number of days VWC was above the
343 CWC threshold (i.e. increasing number of 'wet' days) and total N₂O and manure-derived

344 CH₄ emissions on a per cow per day basis (normalised across regions for differences in
345 stocking rates and days on farm, R²= 0.59, P < 0.001, n=24; Fig. 4).

346 *Restricted grazing scenarios*

347 Adopting DCG for 0 hours per day (i.e. complete removal) on farms with poorly drained
348 soils reduced total N₂O and manure-derived CH₄ emissions by 4 - 12% in Waikato,
349 Manawatu and Southland (Fig. 3). The reduction in N₂O emissions from urine and dung
350 deposition due to cows being completely removed from wet paddocks was only partially
351 offset by increased N₂O emissions from effluent and manure application and CH₄
352 emissions from manure management. Adopting DCG for 13 or 17 hours per day did not
353 result in the same decline in GHG emissions compared to complete removal of cows, with
354 reductions of between 3 - 9% predicted. In contrast, the Canterbury farms showed little
355 change (0 - +2%) in emissions when DCG was implemented (Fig. 3) due to the drier
356 climate (Table 1). The relative impact of DCG when soils were wet on reducing total N₂O
357 and manure-derived CH₄ emissions compared to the baseline varied across regions and
358 increased with the number of 'wet' days. Consequently, DCG was only effective at
359 reducing total N₂O and manure-derived CH₄ emissions on poorly drained soils that had
360 more than ca. 150 'wet' days per year (Fig. 5; includes data from the 10th, 50th and 90th
361 percentile rainfall years).

362 *Insert Figure 3*

363 *Insert Figure 4*

364 *Insert Figure 5*

365 For dairy farms with imperfectly-drained soils, complete removal of cows from wet
366 paddocks in Waikato and Manawatu increased total N₂O and manure-derived CH₄
367 emissions by 6-10% (Fig. 3). This reflects an increase in emissions from manures that
368 more than offset the predicted reductions in paddock-based emissions, indicating pollution
369 swapping. Adopting DCG for 13 or 17 hours on wet days had little effect on total N₂O and
370 manure-derived CH₄ emissions. In Canterbury and Southland, where cows were wintered
371 off in June and July, there was a small increase of 2 - 4% in the total N₂O and manure-
372 derived CH₄ emissions when cows were completely removed from wet paddocks, with
373 very little change (0 - 1%) when DCG was implemented for 13 or 17 hours per day.

374 3.3 Inventory methodology

375 The benefits in reduced GHG emissions achieved from adopting DCG were not apparent
376 when emissions were calculated using the current New Zealand inventory methodology.
377 Firstly, estimated total N₂O and manure-derived CH₄ emissions for farms on imperfectly
378 drained soils were 30-50% greater compared to the refined method (Fig. 6) primarily due
379 to higher paddock-derived N₂O emissions based on a single EF₃ value of 1% for urine
380 compared to lower emissions for imperfectly drained soils based on the VWC and natural
381 logarithmic EF₃ relationship (Fig. 1). Secondly, the current New Zealand inventory
382 methodology assumes 100% of excreta deposited on standoff pads would be stored in
383 'anaerobic lagoons' i.e. effluent pond (Table 2), generating large emissions of CH₄ (0.1095
384 kg CH₄/kg faecal dry matter). In contrast, the refined method assumes most of the excreta
385 is stored as solid manure (Luo et al., 2008), emitting lower rates of CH₄ similar to dung
386 deposition onto pasture (ca. 0.0009 kg CH₄/kg faecal dry matter; Table 2; IPCC, 2006).

387 *Insert Figure 6*

388 3.5 Cost-benefit of adopting DCG when soils were wet to mitigate GHG emissions

389 The cost-benefit of our DCG approach (\$/t CO₂e reduced; Table 4) was calculated for
390 farms on poorly drained soils using modelled total N₂O and manure-derived CH₄ emissions
391 based on the refined inventory approach (Fig. 3) and operating profit (Table 3; sourced
392 from Laurenson et al., submitted). We did not include imperfectly drained soils because
393 there was no reduction in GHG emissions when adopting DCG. For poorly drained soils,

394 the cost:benefit of implementing DCG for 13 hours on wet days in Waikato, Manawatu
395 and Southland ranged from a benefit of \$500 per t CO₂e reduced (Manawatu) to a cost of
396 \$620 per t of CO₂e reduced (Waikato) (Table 4), with higher costs when adopting a longer
397 DCG policy. In contrast to 13 and 17 hour DCG, the cost of completely removing cows
398 from wet paddocks was much greater, at between \$6730 and \$19,000 per t CO₂e reduced.
399 In Canterbury, the small reduction in total GHG emissions for the 13 and 17 hour DCG
400 scenarios substantially increased the cost of adoption (\$14,000-15,000 per t CO₂e reduced;
401 Table 4). The increase in GHG emissions when cows were completely removed from wet
402 paddocks precluded any benefit of this practice, reflecting the relatively low number of wet
403 days in the Canterbury region and the increase in GHG emissions from manure
404 management (Fig. 3).

405

Insert Table 4

406 **4 Discussion**

407 4.1 Method of calculation

408 Our results suggest no benefit can be determined from the proposed DCG for reducing
409 total N₂O and manure-derived CH₄ emissions from dairy farms on either imperfectly
410 drained or poorly drained soils when estimated using the current inventory methodology.
411 Adopting a single EF₃ value of 1% for urine deposited onto soil ignores the influence of
412 soil wetness (and therefore aeration) on microbial-mediated N₂O production (van der
413 Weerden et al., 2012). The refined approach, where urine EF₃ is a function of soil water
414 content, a proxy for soil aeration status, provides a more accurate assessment of the impact
415 of urine deposition on N₂O emissions from wet soils. Another key difference between the
416 two approaches is that the current inventory method assumes any excreta deposited off-
417 paddock is stored in anaerobic lagoons (Ministry for the Environment, 2015), which emit
418 CH₄ at rates much greater than for solid manure (IPCC, 2006). This could inflate the
419 accounting of GHG emissions for farms utilising standoff pads. In practice, excreta
420 deposited onto standoff pads is typically stored as a solid material prior to land application,
421 with negligible amounts of excreta entering ponds. Luo et al. (2008) found that only 4% of
422 the liquid from a standoff pad entered the pond, presumably due to the significant retention
423 of effluent in the woodchip bedding material (Dumont et al., 2012). Inclusion of a second
424 manure management category such as ‘solid storage’ within the inventory methodology
425 would provide a more accurate accounting of emissions from manure deposited onto
426 standoff pads.

427 4.2 Reduction in total N₂O and manure-derived CH₄ emissions

428 The aim of the study was to test if DCG based on a soil water content threshold could
429 reduce farm scale GHG emissions. For poorly drained soils, our DCG approach
430 substantially reduced direct N₂O emissions from excreta deposition when modelled using
431 the refined inventory methodology. The reduction was more than sufficient to offset any
432 increase in N₂O emissions from storage and land application of solid manure. The DCG
433 was most effective at reducing total N₂O emission when cows were completely removed
434 from poorly drained, wet paddocks. In contrast, there was little if any benefit in removing
435 cows from imperfectly-drained soils because the reduction in paddock-based emissions
436 was insufficient to offset a large increase in N₂O emissions associated with storage and
437 land application of solid manure.

438 When including manure-derived CH₄ emissions, implementation of DCG for imperfectly
439 drained soils at the threshold tested will lead to an increase in GHG emissions. Whereas,
440 the CWC used for poorly drained soils led to substantial reductions in total emissions when
441 DCG was implemented, particularly when there are more than 150 ‘wet days’ per year (i.e.
442 VWC > CWC).

443 Previous studies proposed implementation of DCG practices during ‘high risk’ periods
444 such as autumn/winter i.e. a calendar approach (de Klein et al., 2006; Luo et al., 2013) in
445 contrast to our tactical approach. On a poorly drained soil in Southland, limiting cow
446 grazing time to 3 hours per day in autumn (cows wintered off farm for 3 months) reduced
447 total (direct and indirect) on-farm N₂O emissions by 7-11% (de Klein et al., 2006).
448 However, no provision of standoff was made for when soils were wet. Our study showed,
449 for the same region yet cows were wintered off-farm for 2 months only, restricting grazing
450 to 13 hours on wet days reduced N₂O emissions by 9-17% (range of wet, dry and 20-year
451 average rainfall; data not shown). Essentially, our DCG approach produced a greater
452 reduction in total N₂O emissions with less time removed from paddocks compared to de
453 Klein et al.’s (2006) calendar approach. It is also important to note that the earlier study
454 adopted the inventory methodology when modelling N₂O emissions from storage and land
455 application of effluent.

456 Beukes et al. (2011), using the WFM, also adopted a calendar approach when modelling
457 the effectiveness of standoffs as one of five different on-farm GHG mitigation options in
458 the Waikato. They modelled standoff use at 12 hours per day for two months in autumn
459 (March and April) on a dairy farm on a well-drained soil. Total GHG emissions (which
460 included CH₄ enteric fermentation) did not decrease because the reduced N₂O emission
461 from urinary N deposited onto pasture was fully offset by GHG emissions associated with
462 the standoff pad and the application of manure onto pasture. In our study, the Waikato
463 results for an imperfectly drained soil also showed no net decline in total N₂O and manure-
464 derived CH₄ emissions, even though we used soil moisture to derive an EF₃ value and a
465 CWC to remove cows from wet soils. Essentially, on soils that have reasonably good
466 drainage and therefore relatively low N₂O emissions, removing cows from wet soils is
467 likely to result in pollution swapping.

468 Removing cows from wet paddocks will also reduce N leaching, which, in addition to
469 being an indirect source of N₂O emissions (Fig. 3), is a water quality pollutant of major
470 concern in New Zealand (de Klein et al., 2006; Christensen et al., 2012). Our modelled
471 data suggests implementing a 13 hour per day DCG policy could reduce N leaching by up
472 to 13%, providing a co-benefit for its use (data not shown). However, this is a smaller
473 reduction than when compared to complete removal of cows from paddocks during autumn
474 months (following a calendar approach), resulting in ca. 40% reduction in N leaching (de
475 Klein et al., 2006; Vogeler et al., 2013). Therefore, the use of off-paddock facilities will
476 ultimately be dependent on the goals farmers are trying to achieve.

477 4.3 Cost effectiveness of DCG based on a soil water threshold

478 Our analysis suggests adopting a DCG of 13 hours per day on farms with poorly drained
479 soils when soils are wet is most cost-effective in terms of reducing GHG emissions,
480 particularly if the number of ‘wet’ days per year is greater than 150 days. Recently, Vibart
481 et al. (2015) assessed the cost:benefit of a package of mitigation options for Southland
482 dairy farms, which included DCG in addition to other changes including construction of a
483 covered loafing pad and installation of a low rate effluent application. On a dairy farm
484 system similar to that modelled for Southland, this mitigation package cost \$940/t CO₂e
485 reduced relative to the baseline. However, it is difficult to single out the influence of the
486 DCG practice on this value. A more recent analysis showed that employing a standoff for 8
487 hours per day in March and April, with 50% of the herd on a loafing pad in May and June,
488 resulted in a cost of \$2600/t CO₂e reduced (R. Vibart, unpubl. data). Both Vibart’s studies
489 used the OVERSEER[®] model to calculate the GHG emissions (Wheeler et al., 2008),
490 where N₂O emissions from excreta deposition increase with increasing soil water content.
491 While both our refined approach and the OVERSEER predicts urine-derived N₂O
492 emissions in response to soil water content, the former is sensitive to daily changes in soil
493 water content. In contrast, the OVERSEER model operates on a coarser monthly time-step
494 and is therefore less sensitive to rainfall and irrigation events. Adler et al. (2015) used the
495 WFM and New Zealand-specific emission factors to analyse the cost of GHG mitigation

496 strategies for dairy farms in Waikato and Canterbury and found that off-paddock facilities
497 such as standoff pads were a costly alternative compared to other mitigation options such
498 as lower stocking rates and reduced N fertiliser use.

499 Although implementing DCG at 13 hours on wet days was most cost effective for poorly
500 drained soils, the cost:benefit values ranged widely between regional climates, from a
501 desirable benefit of \$500/ t CO₂e reduced in the Manawatu to a cost of \$540-\$620/t CO₂e
502 reduced in Waikato and Southland. The negligible reduction in modelled GHG emissions
503 in Canterbury made DCG financially unviable (estimated cost of \$14,000/t CO₂e reduced).
504 In regions where cows are removed from the dairy platform over the winter months (i.e.
505 ‘wintered off’) such as Canterbury and Southland, the impact of DCG on reducing farm-
506 scale GHG emissions will be limited compared to many North Island regions. This will
507 impact on the financial viability of installing off-paddock facilities such as standoffs with
508 the purpose of reducing GHG emissions due to their associated low return on investment
509 (Adler et al., 2015; Laurenson et al., submitted). Our financial analysis included capital
510 costs associated with construction of the off-paddock facility; our proposed DCG approach
511 will be more financially attractive for farms where off-paddock facilities already exist. It
512 should be noted that our analysis assumed a long-term milk payout of \$6/kg MS
513 (Laurenson et al., submitted).

514 In the current study it was assumed farms were located on a single soil type: future
515 modelling should include farms with mixed soil types (drainage classes). Also, more
516 information on the impact of treading damage and subsequent pasture production, and
517 interaction between damaged soil and urine/dung deposition on N₂O emissions or EFs is
518 needed. Improved understanding of how soil aeration status, relative diffusivity and
519 appropriate methods for measuring or estimating how these parameters affect N₂O
520 emissions is required. This will assist with improving relationships for estimating the
521 impact of grazing and soil damage on emission factors for excreta, fertiliser and manure
522 application to soils.

523 **5 Conclusions**

524 Our analysis suggests that, on farms with poorly drained soils, limiting grazing time to 13
525 hours per day when soils are wet is most cost-effective when aiming to reduce total N₂O
526 and manure-derived CH₄ emissions, particularly if the number of ‘wet’ days (i.e. VWC >
527 CWC) is greater than 150 days. In contrast, there was an increase in emissions for dairy
528 farms on imperfectly-drained soils, suggesting our proposed DCG approach is not suitable
529 for reducing GHG emissions on these soils.

530 **6 Acknowledgements**

531 This project was funded by the New Zealand Government in support of the Livestock
532 Research Group of the Global Research Alliance. We would like to thank Frank Li for
533 conducting initial APSIM modelling and Karren O’Neill for assisting with WFM
534 modelling. We also acknowledge Stewart Ledgard, Mike Hedley, Gerald Cosgrove and
535 Dawn Dalley for providing expert opinion on regional pasture dry matter production
536 levels.

537 **7 References**

538 Adler, A.A., Doole, G.J., Romera, A.J., Beukes, P.C., 2015. Managing greenhouse gas
539 emissions in two major dairy regions of New Zealand: A system-level evaluation.
540 *Agric. Syst.* 135, 1-9.

- 541 Ball, B.C., Cameron, K.C., Di, H.J., Moore, S., 2012. Effects of trampling of a wet dairy
542 pasture soil on soil porosity and on mitigation of nitrous oxide emissions by a
543 nitrification inhibitor, dicyandiamide. *Soil Use Manage.* 28, 194-201.
- 544 Beukes, P.C., Palliser, C.C., Macdonald, K.A., Lancaster, J.A.S., Levy, G., Thorrold, B.S.,
545 Wastney, M.E., 2008. Evaluation of a Whole-Farm Model for pasture-based dairy
546 systems. *J. Dairy Sci.* 91, 2353-2360.
- 547 Beukes, P.C., Gregorini, P., Romera, A.J., 2011. Estimating greenhouse gas emissions
548 from New Zealand dairy systems using a mechanistic whole farm model and
549 inventory methodology. *Anim. Feed Sci. Technol.* 166-167, 708-720.
- 550 Beukes, P. C., Romera, A. J., Clark, D. A., Dalley, D. E., Hedley, M. J., Horne, D. J.,
551 Monaghan, R. M., Laurenson, S., 2013. Evaluating the benefits of standing cows
552 off pasture to avoid soil pugging damage in two dairy farming regions of New
553 Zealand. *NZ J. Agric. Res.* 56, 224-238.
- 554 Butterbach-Bahl, K., Dannenmann, M., 2011. Denitrification and associated soil N₂O
555 emissions due to agricultural activities in a changing climate. *Current Opinion*
556 *Environ. Sustain.* 3, 389-395.
- 557 Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B.,
558 Misselbrook, T., 2011. Manure management: Implications for greenhouse gas
559 emissions. *Anim. Feed Sci. Technol.* 166-167, 514-531.
- 560 Christensen, C. L., Hedley, M. J., Hanly, J. A., Horne, D. J., 2012. Three years of duration-
561 controlled grazing: what have we found? In: *Advanced Nutrient Management:*
562 *Gains from the Past - Goals for the Future.* Occasional Report No. 25
563 <http://flrc.massey.ac.nz/publications.html>. eds L. D. Currie & C. L. Christensen),
564 Fertilizer and Lime Research Centre, Massey University, Palmerston North, New
565 Zealand.
- 566 Cichota R., Snow V.O., Vogeler I., Wheeler D.M., Shepherd M.A. 2012. Describing N
567 leaching from urine patches deposited at different times of the year with a transfer
568 function. *Soil Res.* 50, 694-707.
- 569 Cichota, R., Vogeler, I. Snow, V.O. Webb, T.H., 2013. Ensemble pedotransfer function to
570 derive hydraulic descriptions for New Zealand soils. *Soil Res.* 51, 94-111.
- 571 Clark, H., Brookes, I., Walcroft, A., 2003. Enteric Methane Emissions from New Zealand
572 Ruminants 1990–2001 Calculated Using an IPCC Tier 2 Approach. Report prepared
573 for the Ministry of Agriculture and Forestry by AgResearch and Massey University.
574 Wellington: Ministry of Agriculture and Forestry.
- 575 de Klein, C.A.M., Sherlock, R.R., Cameron, K.C., van der Weerden, T.J., 2001. Nitrous
576 oxide emissions from agricultural soils in New Zealand—a review of current
577 knowledge and directions for future research. *J. Royal Soc. NZ* 31, 543-574.
- 578 de Klein, C.A.M., Barton, L., Sherlock, R.R., Li, Z., Littlejohn, R.P., 2003. Estimating a
579 nitrous oxide emission factor for animal urine from some New Zealand pastoral
580 soils. *Aust. J. Soil Res.* 41, 381-399
- 581 de Klein C.A.M., Li, Z., Sherlock, R.R., 2004. Determination of the N₂O and CH₄
582 emission factors from animal excreta and urea following a winter application in 2
583 regions of New Zealand. Report for MAF Policy, Wellington. Pp. 27.
- 584 de Klein, C.A.M., Smith, L.C., Monaghan, R.M., 2006. Restricted autumn grazing to
585 reduce nitrous oxide emissions from dairy pastures in Southland, New Zealand.
586 *Agric. Ecosyst. Environ.* 112, 192-199.
- 587 Dumont, P.A., Chadwick, D.R., Misselbrook, T.H., Robinson, J.S., Smith, K.A., Sagoo, E.,
588 Camp, V., Murray, R., French, P., Hill, R.A., Scott, A., 2012. Effluent quality and

- 589 ammonia emissions from out-wintering pads in England, Wales and Ireland. *Agric.*
590 *Ecosyst. Environ.* 160, 82-90.
- 591 Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood,
592 J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van
593 Dorland, R., 2007. Changes in atmospheric constituents and in radiative forcing. In:
594 S. Solomon et al (eds.) *Climate change 2007: the physical science basis.*
595 *Contribution of Working Group I to the Fourth Assessment Report of the*
596 *Intergovernmental Panel on Climate Change.* Cambridge University Press,
597 Cambridge, United Kingdom and New York, NY, USA. pp.996.
- 598 Hanigan, M. D., Palliser, C. C., Gregorini, P., 2009. Altering the representation of
599 hormones and adding consideration of gestational metabolism in a metabolic cow
600 model reduced prediction errors. *J. Dairy Sci.* 92, 5043-5056.
- 601 Hewitt, A.E., 2010. *New Zealand Soil Classification.* 3rd edition. Landcare Research
602 Science Series No. 1. Lincoln, Canterbury, New Zealand, Manaaki Whenua Press.
- 603 Holzworth, D.P., Huth, N.I., deVoil, P.G., Zurcher, E.J., Herrmann, N.I., McLean, G.,
604 Chenu, K., van Oosterom, E.J., Snow, V., Murphy, C., Moore, A.D., Brown, H.,
605 Whish, J.P.M., Verrall, S., Fainges, J., Bell, L.W., Peake, A.S., Poulton, P.L.,
606 Hochman, Z., Thorburn, P.J., Gaydon, D.S., Dalgliesh, N.P., Rodriguez, D., Cox,
607 H., Chapman, S., Doherty, A., Teixeira, E., Sharp, J., Cichota, R., Vogeler, I., Li,
608 F.Y., Wang, E., Hammer, G.L., Robertson, M.J., Dimes, J.P., Whitbread, A.M.,
609 Hunt, J., van Rees, H., McClelland, T., Carberry, P.S., Hargreaves, J.N.G.,
610 MacLeod, N., McDonald, C., Harsdorf, J., Wedgwood, S., Keating, B.A., 2014.
611 *APSIM - Evolution towards a new generation of agricultural systems simulation.*
612 *Environ. Modelling and Software,* 62, 327-350.
- 613 Houlbrooke, D. J., Drewry, J. J., Monaghan, R. M., Paton, R. J., Smith, L. C., Littlejohn,
614 R. P., 2009. Grazing strategies to protect soil physical properties and maximise
615 pasture yield on a Southland dairy farm. *NZ J. Agric. Res.* 52, 323-336.
- 616 IPCC, 2006. Emissions from livestock and manure management. In: *Guidelines for*
617 *National Greenhouse Gas Inventories,* Chapter 10. Intergovernmental Panel on
618 Climate Change, 87 pp.[http://www.ipcc-](http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf)
619 [nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf](http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf)
- 620 Générmont, S., Cellier, P., 1997. A mechanistic model for estimating ammonia
621 volatilization from slurry applied to bare soil. *Agric. For. Meteorol.* 88, 145–167.
- 622 Gutser, R., Ebertseder, T., Weber, A., Schraml, M., Schmidhalter, U., 2005. Short-term
623 and residual availability of nitrogen after long-term application of organic fertilizers
624 on arable land. *J. Plant Nutrit. Soil Sci.* 168, 439-446.
- 625 Laubach, J., Heubeck, S., Pratt, C., Woodward, K.B., Guieysse, B., van der Weerden, T.J.,
626 Chung, M.L., Shilton, A.N., Craggs, R.J., 2015. Review of greenhouse gas
627 emissions from the storage and land application of farm dairy effluent. *NZ J. Agric.*
628 *Res.* 58, 203-233.
- 629 Laurenson, S., van der Weerden, T.J., Beukes, P.C., Vogeler, I., Evaluating the economic
630 and production benefits of removing cows from pastures in response to wet soil
631 conditions. Submitted to *Agric. Syst.*
- 632 Li, F.Y., Snow, V.O., Holzworth, D.P., 2011. Modelling seasonal and geographical pattern
633 of pasture production in New Zealand - validating a pasture model in APSIM. *NZ J.*
634 *Agric. Res.* 54, 331-352.
- 635 Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and
636 nitrous oxide production in tilled and nontilled soils. *Soil Sci. Soc. Am. J.* 48, 1267-
637 1272.

- 638 Luo, J., Lindsey, S.B., Ledgard, S.F., 2008. Nitrous oxide emissions from animal urine
639 application on a New Zealand pasture. *Biol. Fert. Soils* 44: 463-470
- 640 Luo, J., Saggar, S., 2008. Nitrous oxide and methane emissions from a dairy farm stand-off
641 pad. *Aust. J. Exp. Agric.* 48, 179-182.
- 642 Luo, J., Ledgard, S.F., Lindsey, S.B., 2013. Nitrous oxide and greenhouse gas emissions
643 from grazed pastures as affected by use of nitrification inhibitors and restricted
644 grazing regime. *Sci. Total Environ.* 456: 107-114.
- 645 Ministry for the Environment, 2015. New Zealand's Greenhouse Gas Inventory 1990-
646 2013. Ministry for the Environment. ISSN 1179-223X. Pp. 450.
- 647 Müller, C., Sherlock, R.R., 2004. Nitrous oxide emissions from temperate grassland
648 ecosystems in the Northern and Southern Hemispheres. *Global Biogeochem. Cycles*
649 18, GB1045 1041-1011.
- 650 Oenema, O., Velthof, G.L., Yamulki, S., Jarvis, S.C., 1997. Nitrous oxide emissions from
651 grazed grassland. *Soil Use Manage.* 13, 288-295.
- 652 Piwowarczyk, A., Giuliani, G., Holden, N.M., 2011. Can soil moisture deficit be used to
653 forecast when soils are at high risk of damage owing to grazing animals? *Soil Use*
654 *Manage.* 27, 255-263.
- 655 Probert, M.E., Dimes, J.P., Keating, B.A., Dalal, R.C., Strong, W.M., 1998. APSIM's
656 water and nitrogen modules and simulation of the dynamics of water and nitrogen in
657 fallow systems. *Agric. Syst.* 56, 1-28.
- 658 Reay, D.S., Davidson, E.A., Smith, K.A., Smith, P., Melillo, J.M., Dentener, F., Crutzen,
659 P.J., 2012. Global agriculture and nitrous oxide emissions. *Nature climate change*, 2,
660 410-416.
- 661 Romera, A.J., McCall, D.G., Lee, J.M., Agnusdei, M.G., 2009. Improving the McCall
662 herbage growth model. *NZ J. Agric. Res.* 52, 477-494.
- 663 Ryden, J.C., Ball, P.R., Garwood, E.A., 1984. Nitrate leaching from grassland. *Nature* 311,
664 50-53.
- 665 Selbie, D.R., Buckthought, L.E., Shepherd, M.A., 2015a. The challenge of the urine patch
666 for managing nitrogen in grazed pasture systems. In: Donald L S ed. *Adv. Agron.*
667 129, 229-292.
- 668 Sherlock, R.R., de Klein C.A.M., Li, Z., 2003a. Determination of the N₂O and CH₄
669 emission factors from animal excreta, following a spring application in 3 regions of
670 New Zealand. Report for MAF Policy, Wellington. Pp. 28
- 671 Sherlock, R.R., de Klein C.A.M., Li, Z., 2003b. Determination of the N₂O and CH₄
672 emission factors from animal excreta, following a summer application in 3 regions
673 of New Zealand. Report for MAF Policy, Wellington. Pp. 27
- 674 Snow, V.O., Huth, N., 2004. The APSIM-Micromet module. HortResearch Internal Report
675 No. 2004/12848, HortResearch, Palmerston North, New Zealand.
- 676 Soil Survey Staff, 1998. Keys to soil taxonomy. United States Department of Agriculture,
677 Washington, DC.
- 678 Tait, A., Henderson, R., Turner, R., Zheng, X. 2006. Thin-plate smoothing spline
679 interpolation of daily rainfall for New Zealand using a climatological rainfall
680 surface. *Internat. J. Climatol.* 26, 2097-2115.
- 681 van der Weerden, T.J., Luo, J., de Klein, C.A.M., Hoogendoorn, C.J., Littlejohn, R.P., Rys,
682 G.J., 2011. Disaggregating nitrous oxide emission factors for ruminant urine and
683 dung deposited onto pastoral soils. *Agric. Ecosyst. Environ.* 141, 426-436.

- 684 van der Weerden, T.J., Kelliher, F.M., de Klein, C.A.M., 2012. Influence of pore size
685 distribution and soil water content on nitrous oxide response curves. *Soil Res.* 50,
686 125-135.
- 687 van der Weerden, T.J., Manderson, A., Kelliher, F.M., de Klein, C.A.M., 2014. Spatial and
688 temporal nitrous oxide emissions from dairy cattle urine deposited onto grazed
689 pastures across New Zealand based on soil water balance modelling. *Agric. Ecosyst.*
690 *Environ.* 189, 92-100.
- 691 van der Weerden, T.J., Cox, N., Luo, J., Di, H.J., Podolyan, A., Phillips, R.L., Saggar, S.,
692 de Klein, C.A.M., Ettema, P., Rys, G., 2016. Refining the New Zealand nitrous
693 oxide emission factor for urea fertiliser and farm dairy effluent. *Agric. Ecosyst.*
694 *Environ.* 222, 133-137.
- 695 van Groenigen, J.W.; Velthof, G.L.; van der Bolt, F.J.E.; Vos, A.; Kuikman, P.J., 2005.
696 Seasonal variation in N₂O emissions from urine patches: effects of urine
697 concentration, soil compaction and dung. *Plant and Soil* 273, 15-27.
- 698 Vibart, R., Vogeler, I., Dennis, S., Kaye-Blake, W., Monaghan, R., Burggraaf, V.,
699 Beautrais, J., Mackay, A., 2015. A regional assessment of the cost and effectiveness
700 of mitigation measured for reducing nutrient losses to water and greenhouse gas
701 emissions to air from pastoral farms. *J. Environ. Manage.* 156, 276-289.
- 702 Vogeler, I., Beukes, P., Burggraaf, V., 2013. Evaluation of mitigation strategies for nitrate
703 leaching on pasture-based dairy systems. *Agric. Syst.* 115, 21-28.
- 704 Webb, J., Sørensen, P., Velthof, G., Amon, B., Pinto, M., Rodhe, L., Salomon, E.,
705 Hutchings, N., Burczyk, P., Reid, J., 2013. An Assessment of the Variation of
706 Manure Nitrogen Efficiency throughout Europe and an Appraisal of Means to
707 Increase Manure-N Efficiency. *Adv. Agron.* 119, 371-441.
- 708 Wheeler, D.M., Ledgard, S.F., de Klein, C.A.M., 2008. Using the OVERSEER nutrient
709 budget model to estimate on-farm greenhouse gas emissions. *Austral. J. Exp. Agric.*
710 48, 99-103.
- 711 Whitehead, D.C., 1995. *Grassland Nitrogen*. CAB International, Wallingford, UK.
- 712

713 Figure Captions

714

715 Figure 1. Relationship between modelled VWC averaged over 20 days from time of urine
716 deposition and natural log of measured dairy cattle urine N₂O emission factor (ln EF₃, %)

717

718 Figure 2. Comparison of direct N₂O emissions (kg N₂O-N/ha/yr) from urine deposition on
719 grazed paddocks when adopting DCG for 0 (i.e. complete removal), 13 or 17 hours per day
720 compared to 21 hours per day (baseline) when soil moisture > CWC for an imperfectly-
721 drained (●) and poorly drained (□) soil, calculated using a refined methodology based on
722 soil moisture content (left) and the current New Zealand inventory methodology (right).
723 Values modelled for the 50th percentile rainfall year.

724

725 Figure 3. Total N₂O and manure-derived CH₄ emissions ('Total emissions', kg
726 CO₂e/ha/year) from baselines and 3 duration controlled grazing scenarios (0 (i.e. complete
727 removal), 13 and 17 hours' grazing per day when soil moisture > CWC) for an
728 imperfectly-drained and poorly drained soil in four regions (a: Waikato, b: Manawatu, c:
729 Canterbury, d: Southland). Values modelled for the 50th percentile rainfall year using a
730 refined inventory methodology.

731

732 Figure 4: Relationship between number of days VWC > CWC and total N₂O and manure-
733 derived CH₄ emissions (kg CO₂e/cow/day) for baseline (i.e. cows not removed from wet
734 paddocks). Values modelled for two drainage classes by four regions by three years (10th,
735 50th and 90th percentile rainfall years) (n=24) using a refined inventory methodology.

736

737 Figure 5: Relationship between number of days $VWC > CWC$ and reduction in total N_2O
738 and manure-derived CH_4 emissions ($kg\ CO_2e/cow/day$) when duration controlled grazing
739 implemented for 0, 13 or 17 hours per day for poorly drained soils only. Values modelled
740 for two drainage classes by four regions by three years (10th, 50th and 90th percentile
741 rainfall years) (n=24) using a refined inventory methodology.

742

743 Figure 6: Comparison of total N_2O and manure-derived CH_4 emissions ('Total emissions',
744 $kg\ CO_2e/ha/year$) based on current New Zealand inventory methodology (\square) and refined
745 methodology (\blacksquare) from baseline dairy farms and when DCG implemented for 0 (i.e. no
746 grazing), 13 and 17 hours per day when soil moisture $> CWC$ on an imperfectly-drained
747 and poorly drained soil in four regions (a: Waikato, b: Manawatu, c: Canterbury, d:
748 Southland). Values modelled for the 50th percentile rainfall year. Black bars correspond to
749 the total emissions reported in Fig 3.

Table 1. Details of regions, climates and dairy farm production values.

Region	Location	Coordinates	Year ^A	Relative Rainfall ^A	Actual Rainfall (mm)	Typical pasture production (t DM/ha/yr)		Stocking rate (cows/ha)		No. days above CWC ^B	
						Imperfectly drained	Poorly drained	Imperfectly drained	Poorly drained	Imperfectly drained	Poorly drained
Waikato	Hamilton	37.775 S, 175.325 E	2013-14	10 th percentile	873	12.0	12.0	2.95	2.95	83	201
			2012-13	50 th percentile	1097	14.5	14.0			111	181
			2010-11	90 th percentile	1439	17.5	17.0			141	243
Manawatu	Palmerston North	40.375 S, 175.625 E	2007-08	10 th percentile	845	10.0	9.0	2.95	2.95	87	204
			1996-97	50 th percentile	1000	12.8	11.3			102	213
			1995-96	90 th percentile	1220	14.0	11.3			156	270
Canterbury	Lincoln	43.625 S, 172.475 E	1998-99	10 th percentile	471 (+375 ^C)	17.0	17.0	3.9	3.9	15 ^D	142 ^D
			2007-08	50 th percentile	631 (+325 ^C)	18.5	18.5			25 ^D	152 ^D
			2008-09	90 th percentile	879 (+275 ^C)	20.0	20.0			41 ^D	147 ^D
Southland	Winton	46.125 S, 168.325 E	2002-03	10 th percentile	823	9.7	15.1	2.75	3.15	50 ^D	135 ^D
			1999-2000	50 th percentile	898	11.0	17.4			32 ^D	145 ^D
			1996-97	90 th percentile	1017	12.5	20.0			71 ^D	172 ^D

^A Year was chosen based on the 10th, 50th and 90th percentile rainfall experienced in each region between 1995 and 2014; ^B CWC = critical water content; ^C Values in brackets refer to irrigation applied (mm) to supplement rainfall (applied when soil water deficit of 20-25 mm present), ^D Excludes June, July and early August, when cows were wintered off farm.

Table 2: Calculation of total greenhouse gas emissions (excluding enteric fermentation) for modelled dairy farms using New Zealand IPCC inventory methodology and improvements to methodology.

Component of calculation	Code	New Zealand IPCC inventory methodology	Potential improvements to inventory methodology	Comments
N ₂ O emission factor for urine (kg N ₂ O-N/kg N)	EF _{3PRP}	0.01	Dependent on soil water content.	Based on relationship between EF _{3PRP} and VWC (Fig. 1).
N ₂ O emission factor for dung (kg N ₂ O-N/kg N)	EF _{3PRP DUNG}	0.0025	NC ^A	
N ₂ O emission factor for urea fertiliser (kg N ₂ O-N/kg N)	EF _{1 UREA}	0.0048	0.006	van der Weerden et al. (2016)
Fraction of N _{EX} or urea fertiliser N leached (kg NO ₃ -N/kg N)	Frac _{LEACH}	0.07	Modelled using APSIM	Uses local climate and soil data
Fraction of FDE N leached (kg NO ₃ -N/kg N)	Frac _{LEACH FDE}	0.07	NC	
N ₂ O emission factor for N leached (kg N ₂ O-N/kg N)	EF ₅	0.0075	NC	
Fraction of N _{EX URINE} lost through NH ₃ volatilisation (kg NH ₃ -N/kg N)	Frac _{GASM URINE}	0.10	Modelled using APSIM	Uses local climate and soil data
Fraction of N _{EX DUNG} lost through NH ₃ volatilisation (kg NH ₃ -N/kg N)	Frac _{GASM DUNG}	0.10	NC	
Fraction of urea fertiliser lost through NH ₃ volatilisation (kg NH ₃ -N/kg N)	Frac _{GASF}	0.10	Modelled using APSIM	Uses local climate and soil data
N ₂ O emission factor for NH ₃ volatilisation (kg N ₂ O-N/kg N)	EF ₄	0.01	NC	
N ₂ O emission factor effluent storage in uncovered anaerobic lagoon (kg N ₂ O-N/kg N)	EF _{3(S AL)}	0	NC	
N ₂ O emission factor excreta deposited onto standoff pad (=solid storage). (kg N ₂ O-N/kg N)	EF _{3(SS)}	Not considered; therefore treated all excreta on standoff pad as EF _{3(S AL)} (= 0)	0.0001	Luo and Sagar (2008)
Fraction of effluent N leached during storage in uncovered anaerobic lagoon (kg NO ₃ -N/kg N)	Frac _{LEACH MS}	0	NC	

Fraction of effluent N lost as NH ₃ during storage (kg NH ₃ -N/kg N)	Frac _{GASMS} FDE	0.35	NC	
Fraction of stored effluent in anaerobic lagoon lost during storage as gaseous N (kg N/kg N)	Frac _{LOSSMS} FDE	0.35 ^B	NC	2006 IPCC guidelines, Chapter 10, Table 10.23 (IPCC, 2006)
Fraction of stored effluent applied to land, adjusted for N lost during manure management system (kg N/kg N)	Frac _{NEX} EFFAPP	1 - Frac _{LOSSMS} FDE = 0.65	NC	2006 IPCC guidelines, Chapter 10, Equation 10.34 (IPCC, 2006)
Fraction of excreta N from standoff pad MM lost as NH ₃ (kg NH ₃ -N/kg N)	Frac _{GASMS} SO	Not considered; therefore treated as effluent (0.35)	0.30	Assumed Standoff pad = 'Solid Storage' MM (2006 IPCC guidelines, Chapter 10, Table 10.18); Frac _{GASMS} given in Table 10.22 (IPCC, 2006)
Fraction of standoff pad N excreta entering anaerobic lagoon (kg N/kg N)	Frac _{NEX} SO→AL	Not considered; therefore treated as effluent (1.0)	Assumed 0.04	Luo et al. (2008)
Fraction of standoff pad excreta applied to land, adjusted for N lost during manure management system (kg N/kg N)	Frac _{NEX} SO APP	Not considered; therefore assumed same as FDE: 1 - Frac _{LOSSMS} FDE = 0.65	Assume 1 - (EF ₃ S SS + Frac _{LOSSMS} SO) = 0.70	
N ₂ O emission factor for farm dairy effluent (kg N ₂ O-N/kg N)	EF ₁ FDE	0.01	0.003	van der Weerden et al. (2016)
N ₂ O emission factor for standoff pad manure applied to land (kg N ₂ O-N/kg N)	EF ₁ SO	0.01	NC	
Fraction of N _{EX} , applied FDE or applied standoff pad manure lost through NH ₃ volatilisation (kg NH ₃ -N/kg N)	Frac _{GASM}	0.10	NC	
Fraction of standoff pad manure N leached (kg NO ₃ -N/kg N)	Frac _{LEACH} SO	Not considered; therefore treated as effluent (0.07) (consistent with other N loss pathways)	0	Assumed Standoff pad manure applied to land under good practice. Also, less mobile form of N.
CH ₄ emissions for N _{EX} DUNG deposited onto pasture (kg CH ₄ /kg FDM ^C)	CH ₄ PRP	0.00098	NC	
CH ₄ emissions from stored effluent in	CH ₄ MM	0.1095	NC	

anaerobic lagoons (kg CH ₄ /kg FDM stored)				
CH ₄ emissions from excreta deposited onto standoff pads (= solid storage) (kg CH ₄ /cow/year)	CH ₄ _{MM}	Not considered, therefore treated as effluent FDM entering anaerobic lagoons.	VS = 3.5 kg/cow/day; B ₀ = 0.24; MCF = 4% (see footnote for description).	2006 IPCC guidelines, Chapter 10, Equation 10.23; Table 10.A4 (IPCC, 2006)

^A NC = no change to NZ inventory methodology; ^B NZ inventory assumes all gaseous N losses from anaerobic lagoon are as NH₃, with nil N₂ emissions (MPI, pers. comm.2015); ^C FDM = Faecal dry matter; VS = volatile solids; B₀ = maximum methane producing capacity for manure produced by cattle; MCF = methane conversion factor.

Table 3. Change in dairy operating profit (DOP; \$/ha/year) for duration controlled grazing (DCG) scenarios (0, 13 and 17 hours grazing per day when soils are wet) for dairy farms with poorly drained soils. Negative values indicate a reduction in DOP, positive values indicate an increase in DOP. Values shown are for the 50th percentile rainfall year (source: Laurenson et al. submitted.).

DCG (hours/day when VWC > CWC)	Region			
	Waikato	Manawatu	Canterbury	Southland
0 (no grazing)	-\$2,539	-\$2,299	-\$1,843	-\$1,522
13	-\$148	+\$81	-\$155	-\$142
17	-\$222	-\$91	-\$178	-\$139

Table 4. Cost:benefit of contrasting duration controlled grazing (DCG) scenarios when poorly drained soils were wet to reduce GHG emissions (\$ per t CO₂e reduced). Negative values indicate a cost, positive values indicate a benefit; for Canterbury, 0 hours excluded due to emissions increasing relative to baseline. Values shown are based on 20-year average rainfall.

DCG scenarios (hours/day when VWC > CWC)	Region			
	Waikato	Manawatu	Canterbury	Southland
0 (no grazing)	-\$6,730	-\$19,000		-\$7,320
13	-\$620	+\$500	-\$14,130	-\$540
17	-\$1,520	-\$900	-\$14,800	-\$1,340

Fig. 1.

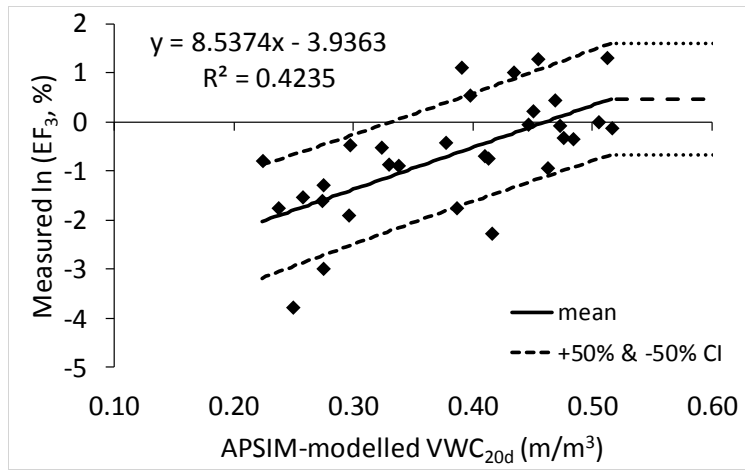
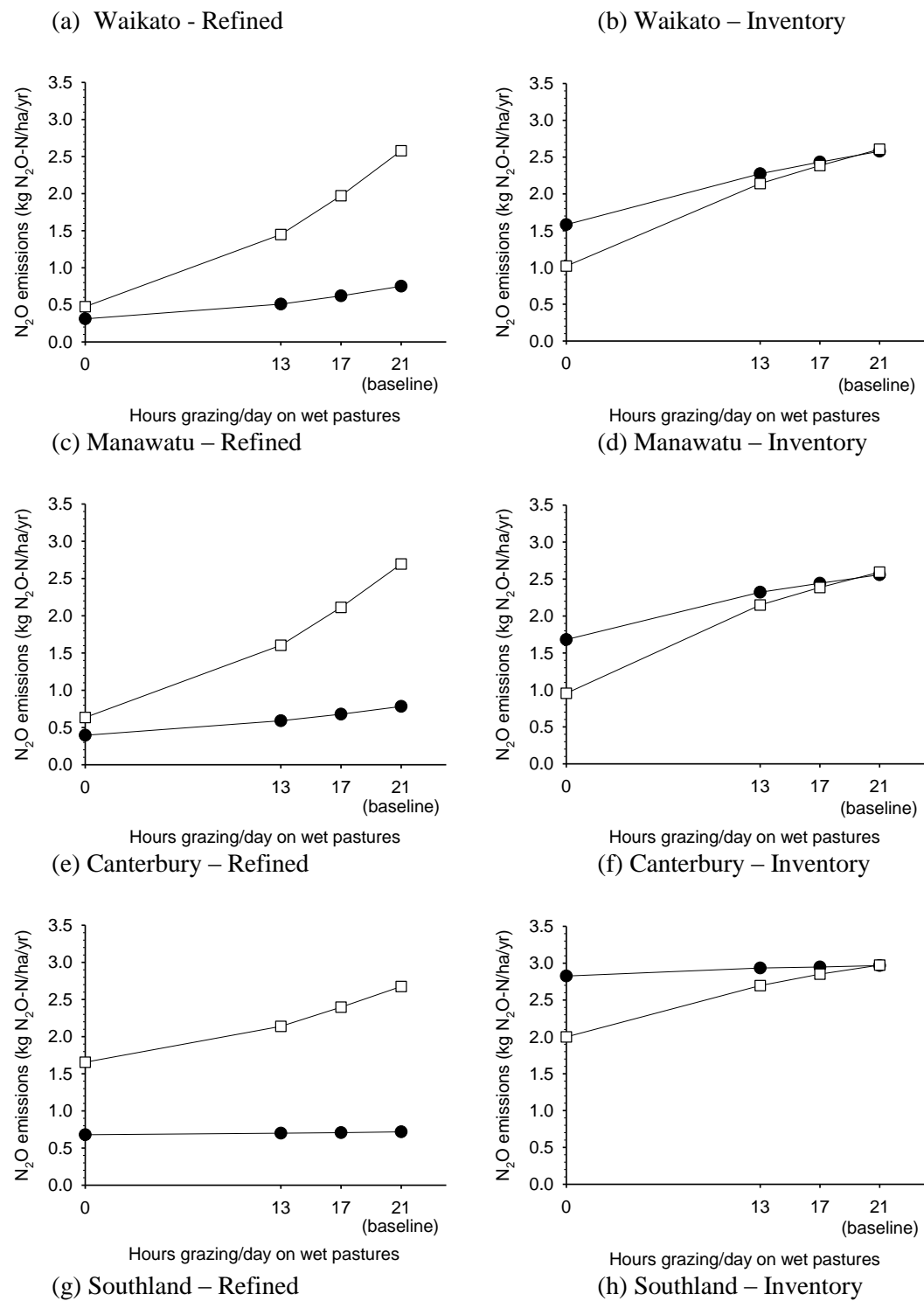


Fig. 2.



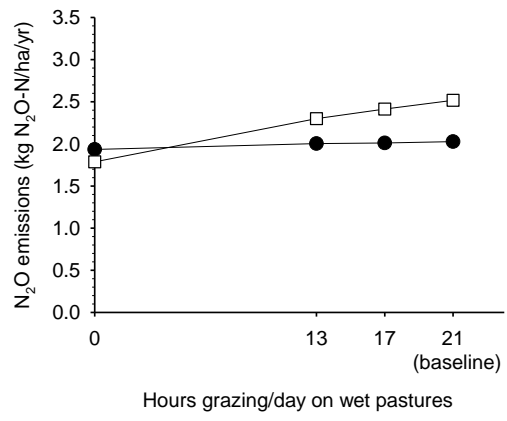
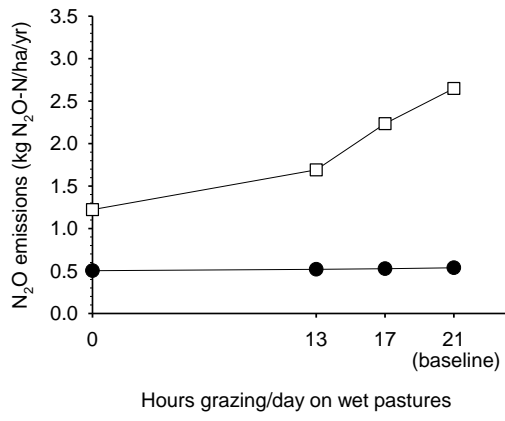


Fig. 3.

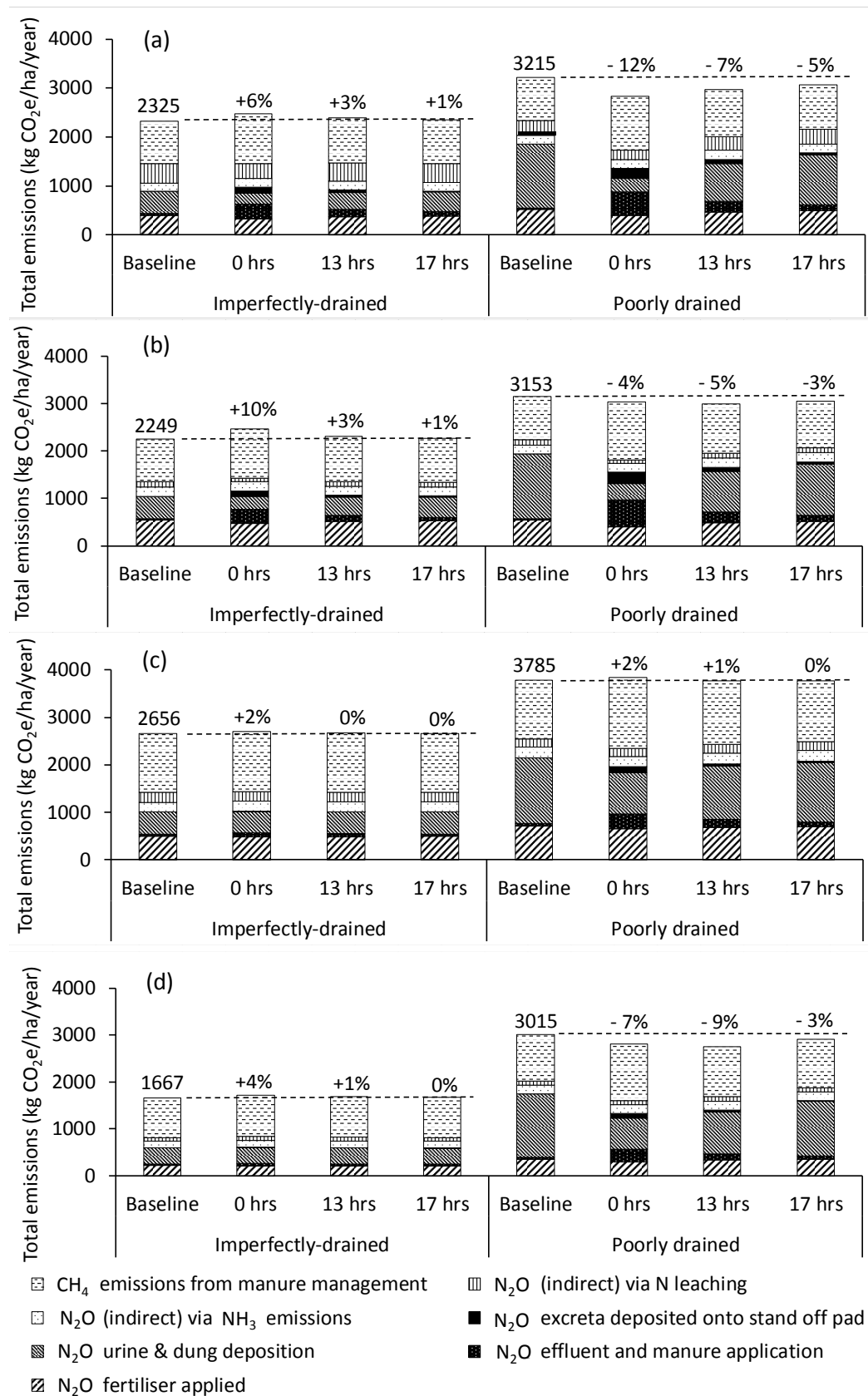


Fig. 4.

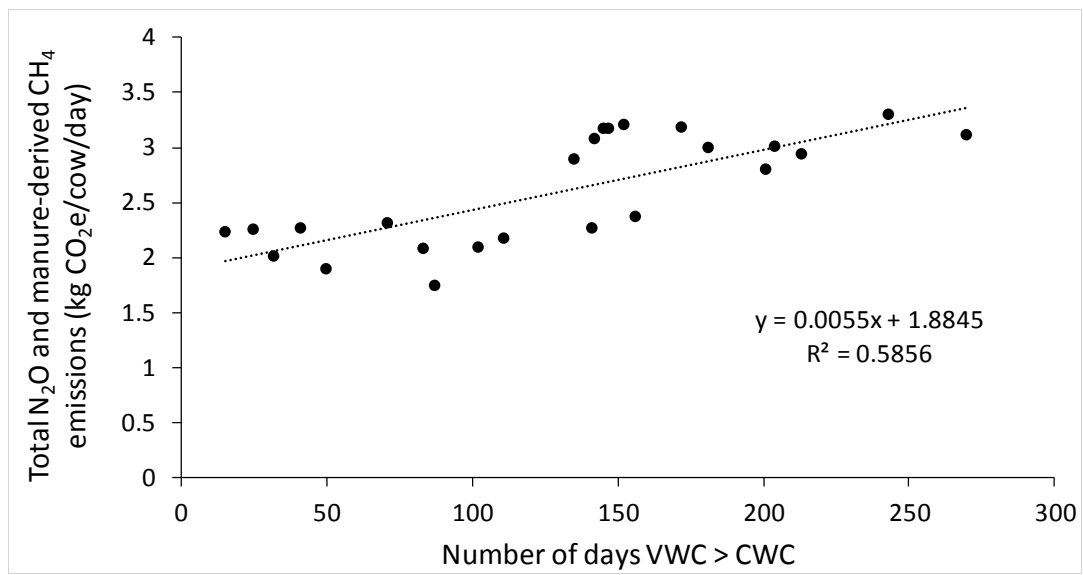


Fig. 5.

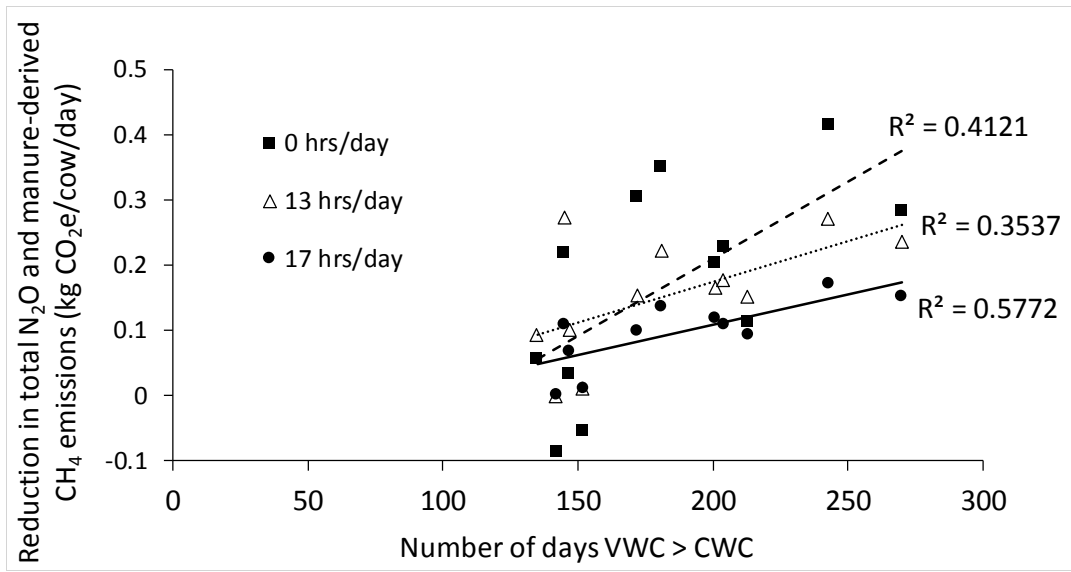


Fig. 6.

