

1 **How do farm models compare when estimating greenhouse gas emissions**
2 **from dairy cattle production?**

3 N.J.Hutchings¹, Ş. Özkan Gülzari ^{2,a}, M. de Haan ³ and D. Sandars ⁴

4

5 ¹ *Department of Agroecology, Aarhus University, Blichers Allé 20, P.O. Box 50, Tjele,*
6 *8830 Denmark*

7 ² *Department of Animal and Aquacultural Sciences, Faculty of Veterinary Medicine*
8 *and Biosciences, Norwegian University of Life Sciences (NMBU), P.O. Box 5003, Ås,*
9 *1430 Norway*

10 ³ *Wageningen UR, Livestock Research, P.O. Box 338 Wageningen, 6700AH, The*
11 *Netherlands*

12 ⁴ *School of Water, Energy, and Environment, Cranfield University, Bedford, MK43*
13 *0AL UK*

14 ^a *Present address: Norwegian Institute of Bioeconomy Research, P.O. Box 115, Ås*
15 *1431 Norway*

16

17 Corresponding author: Nicholas Hutchings. Email: nick.hutchings@agro.au.dk

18

19 Short title: Comparing dairy cattle farm model greenhouse emissions

20

21 **Abstract**

22 The European Union (EU) Effort Sharing Regulation will require a 30% reduction in
23 greenhouse gas (GHG) emissions from the sectors not included in the European
24 Emissions Trading Scheme, including agriculture. This will require the estimation of
25 baseline emissions from agriculture, including dairy cattle production systems. To

26 support this process, four farm-scale models were benchmarked with respect to
27 estimates of greenhouse gas (GHG) emissions from six dairy cattle scenarios; two
28 climates (cool/dry and warm/wet) x two soil types (sandy and clayey) x two roughage
29 production systems (grass only and grass/maize). The milk yield per cow (7000 kg
30 Energy-corrected milk (ECM) year⁻¹), follower:cow ratio (1:1), manure management
31 system and land area were standardised for all scenarios. Potential yield and
32 application of available N in fertiliser and manure were standardised separately for
33 grass and maize. Significant differences between models were found in GHG
34 emissions at the farm-scale and for most contributory sources, although there was no
35 difference in the ranking of source magnitudes. The difference between the models
36 with the lowest and highest GHG emission intensities, averaged over the six
37 scenarios (0.08 kg CO₂e (kg ECM)⁻¹), was similar to the difference between the
38 scenarios with the lowest and highest emission intensities (0.09 kg CO₂e (kg ECM)⁻¹),
39 averaged over the four models, indicating that if benchmarking is to contribute to
40 the quality assurance of emission estimates, there needs to be further discussion
41 between modellers, and between modellers and those with expert knowledge of
42 individual emission sources, concerning the nature and detail of the algorithms
43 needed. Even though key production characteristics were standardised in the
44 scenarios, there were still significant differences between models in the milk
45 production ha⁻¹ and the amounts of N fertiliser and concentrate feed imported. This
46 was because the models differed both in their description of biophysical
47 responses/feedback mechanisms and in the extent to which management functions
48 were internalised. This shows that benchmarking farm models for dairy cattle
49 systems will be more difficult than for those agricultural production systems where
50 feedback mechanisms are less pronounced.

51

52 **Keywords:** cattle, farm, model, greenhouse gas

53

54 **Implications**

55 If farm scale models of GHG emissions are to be useful in the more stringent
56 regulatory environment in Europe, there needs to be further discussion between
57 modellers, and between modellers and those with expert knowledge of individual
58 emission sources, concerning the nature and detail of the algorithms used.

59 Benchmarking can help maintain the quality of such models but feedback
60 mechanisms exist within ruminant livestock systems that will make this more difficult
61 than for other agricultural production systems.

62

63 **Introduction**

64 Globally, the livestock sector accounts for 14.5% of human-caused greenhouse gas
65 emissions (GHG), producing 7.1 Gt of carbon dioxide equivalent (CO_{2e}) emissions
66 year⁻¹, of which dairy farming contributes about 20% (Hagemann *et al.*, 2012).
67 European dairy production is about 150 million tonnes of milk (European Dairy
68 Association, 2016) and accounts for about 14% of the value of all agricultural
69 production (https://ec.europa.eu/agriculture/milk_en). However, it also accounts for
70 about one third of GHG emissions from the European livestock sector (Bellarby *et al.*,
71 2013) The sources of direct GHG emissions are methane (CH₄) from enteric
72 fermentation and manure management and nitrous oxide (N₂O) from manure
73 management and the soil. In addition, there are indirect GHG emissions in the form
74 of N₂O, resulting from the nitrification and partial denitrification of reduced forms of
75 nitrogen (N) that occur off-farm, either as a result of the atmospheric deposition of N

76 from ammonia (NH₃) volatilization from manure management and the soil, or from
77 nitrate (NO₃⁻) leaching from the soil (IPCC, 2006).

78 Hitherto, there has been limited pressure to reduce GHG emissions from agriculture,
79 although there is increased interest from the food retail sector concerning their GHG
80 emissions and that of their supply chains (e.g. Tesco PLC, 2016). However, the
81 European Union (EU) is currently in the process of supplementing its Effort Sharing
82 Decision (European Commission, 2009) with an Effort Sharing Regulation (ESR;
83 Erbach, 2016) that by 2030, will reduce by 30% the GHG emissions from the sectors
84 not included in the European Emissions Trading Scheme (agriculture, transport,
85 buildings, small industry and waste). The agreement will place a heavier burden on
86 the wealthier Member States and impose national Annual Emission Allocations but
87 will allow some flexibility concerning the distribution of reduction burden between
88 sectors and allow limited transfer or trading of Annual Emission Allocations. How the
89 ESR will be implemented in individual Member States is unclear, including the
90 proportion of the emission reduction allocated to agriculture and the extent to which
91 there is the ability and willingness to utilise the flexibility mechanisms. However,
92 since the ESR contains reduction targets for EU member states that range from 0 to
93 40%, significant reductions seem likely to be demanded from agriculture, especially
94 for more wealthy Member States with large agricultural sectors. The extent to which
95 Member States choose to allocate reduction targets to individual agricultural
96 production sectors or to individual farms has also yet to be decided.

97 Measurements of GHG emissions are not currently available at the farm scale and
98 given the technical and financial challenges (Brentrup *et al.*, 2000, McGinn, 2006) it
99 seems unlikely that this situation will change in the near future. Consequently,
100 estimates of GHG emissions from agriculture for the farm scale and above are

101 obtained by modelling. Ruminant livestock farms in general, and dairy cattle farms in
102 particular, typically rely heavily on on-farm crop production to supply animal feed.
103 This leads to a substantial internal cycling of nutrients (Jarvis *et al.*, 2011), feedback
104 effects between farm components (livestock, manure management etc.) and difficulty
105 in obtaining the information concerning feed intake necessary to calculate the major
106 sources of GHG emissions. As a consequence, it is appropriate to rely on whole-farm
107 systems models (Crosson *et al.*, 2011).

108 A number of whole-farm cattle systems models have been developed to address this
109 situation (Del Prado *et al.*, 2013, Kipling *et al.*, 2016). At present, these models have
110 mainly been used for exploratory purposes e.g. Vellinga *et al.* (2011), for which
111 plausibility is an adequate criteria for the form of response functions and the quality
112 of inputs and parameters. Exploration will remain a useful function but in the future,
113 farm-scale models will also need to operate within an environment in Europe in which
114 there is regulatory or commercial pressure to reduce emissions and in which the
115 quality of emission inventories at all scales is likely to be subject to increased
116 scrutiny. Comparing the results from different models when used to simulate
117 standard scenarios (benchmarking) can contribute to the quality assurance or review
118 processes.

119 In order to achieve target-based reductions in GHG emissions, such as those
120 proposed in the ESR, there is a need to establish baseline emissions i.e. emissions
121 prior to the implementation of abatement measures. In the study reported here, we
122 quantify the differences between four farm-scale models in the GHG emissions using
123 six standard scenarios of dairy cattle production and identify the differences in the
124 structure and function of the models that give rise to these differences.

125

126 **Material and methods**

127 The models used were DairyWise, developed in The Netherlands (Schils *et al.*,
128 2007), FarmAC, developed as part of an EU project (Hutchings and Kristensen,
129 2015), HolosNor, developed in Norway (Bonesmo *et al.*, 2012), and SFARMMOD,
130 developed in the United Kingdom (Annetts and Audsley, 2002). DairyWise and
131 HolosNor are specifically dedicated to dairy farming whereas FarmAC and
132 SFARMOD can simulate a wider range of farm types. The choice of models used
133 depended on who could obtain funding via the Modelling European Agriculture with
134 Climate Change for Food Security (MACSUR) project (www.macsur.eu). A brief
135 background to each model used in the current comparison study is given in
136 Supplementary Material. The order of the models is alphabetical with no intention to
137 rank them. Emissions are expressed in kg CO₂e year⁻¹ and CO₂e (kg ECM⁻¹; i.e.
138 emissions intensity). The models varied in the GHG sources included. Not all models
139 could simulate off-farm GHG emissions, such as pre- or post-chain emissions. Nor
140 could all models simulate emissions associated with the use of farm machinery or the
141 sequestration of carbon (C) in the soil, so these were omitted from the comparison.
142 Global warming potentials (GWP) of CH₄ and N₂O are 28 and 265 times higher than
143 that of CO₂, respectively, for a given 100 year time horizon (Myhre *et al.*, 2013).

144

145 *Scenarios*

146 Each model simulated eight scenarios within a factorial design consisting of two
147 climates, two soil types, and two feeding systems. The two climates were cool with
148 moderate rainfall (Wageningen, The Netherlands) and warm with high rainfall
149 (Santander, Spain). The Cool climate had a mean annual temperature of 9.6 °C and
150 a mean annual precipitation of 757 mm. The Warm climate had a mean annual

151 temperature 14.3 °C and a mean annual precipitation of 1268 mm. The
152 characteristics of the Sandy soil were 60% sand, 10% silt, 30% clay and the Clayey
153 soil were 10% sand, 45% silt, 45% clay. For both soil types, the pH >6, <7.5 and soil
154 depth was 1 metre. For HolosNor, the maximum permissible clay content allowed by
155 the model (35%) was used (A. O. Skjelvåg, Ås, 2016, personal communication).

156 The choice of scenarios was intended to provoke noticeable responses from the
157 models whilst remaining within the range of conditions for European dairy production.
158 The choice of climates was also determined by the need to access advice concerning
159 climate-related farm management information. Grass has an energy:protein ratio that
160 is sub-optimal for effective utilisation of the protein for milk production, so must be
161 supplemented with an energy-rich feed when formulating diets (Özkan and Hill,
162 2015). This is commonly provided using either an imported cereal or on-farm maize
163 silage, so two cropping systems were simulated, one consisting of grass only and
164 other of grass and maize silage.

165 The interested partners agreed a set of standardised farm structure and
166 management characteristics and parameters (Table 1). The emission intensity of milk
167 production decreases with increasing annual milk production per cow (Casey and
168 Holden, 2005, Gerber *et al.*, 2011), so it was necessary to standardise this factor. To
169 avoid excessive externalising of GHG emissions through high imports of energy
170 concentrates and to be relevant for as much of European dairy production as
171 possible, we chose to simulate a production system with a moderate production of
172 7000 kg ECM cow⁻¹ year⁻¹, rather than one designed to be typical for the two climates
173 chosen. Typical farms in the relevant regions of Netherlands and Spain would
174 produce about 7400 and 8400 kg ECM cow⁻¹ year⁻¹.

175

176 Table 1 here

177

178 Complete standardisation of scenarios was not possible as all models required
179 additional model-specific inputs or parameters. To internalize model responses, the
180 exchange of material with off-farm systems was minimized. This meant that within
181 realistic constraints (e.g. maintaining a realistic balance between energy and protein
182 in cattle diets), the amount of imported animal feed and manure and the export of
183 silage and manure was minimised. Since the milk yield per cow, the weight of the
184 mature dairy cows and the number of young stock per mature dairy cow were
185 standardised, the number of livestock that could be carried on the farm was
186 determined by each model's prediction of (i) the diet necessary to achieve the
187 specified milk yield and growth of immature livestock; and (ii) the capacity of the farm
188 to produce roughage feed. HolosNor required the number of animals as an input;
189 therefore, the number of animals in each scenario was inputted to HolosNor from
190 FarmAC.

191 The statistical significance of the differences between models for the selected
192 management variables and the estimated GHG emissions was determined using the
193 Friedman test (Friedman, 1940), followed by the post-hoc Nemenyi test (Nemenyi,
194 1963). The analysis was undertaken using the `Friedman.test` and
195 `posthoc.friedman.nemenyi.test` function from the PMCMR package (Pohlert, 2014) of
196 R programming language.

197

198 **Results**

199 *Differences between scenarios*

200 The emission intensities for the different scenarios, averaged across models, are
201 shown in Table 2. There were systematic differences between the grass only and
202 grass/maize systems, with the grass only system required more concentrate feed,
203 carried a higher livestock number and received more N fertiliser. The enteric CH₄
204 emissions were lower for the grass/maize system than the grass only. Manure CH₄
205 emissions varied little across scenarios whereas manure N₂O emission tended to be
206 lower in the warm climate. The field N₂O emissions were similar for all scenarios.
207 Nitrous oxide emissions associated with NH₃ volatilisation were slightly lower for the
208 grass/maize system. Nitrous oxide emissions associated with NO₃⁻ leaching were
209 greatest for the sandy soil than the clayey soil. The total GHG emission intensity was
210 around 4% greater for the grass only system (1.11 kg CO₂e (kg ECM)⁻¹) than for the
211 grass/maize (1.07 kg CO₂e (kg ECM)⁻¹), and greater for the cool climate (1.12 kg
212 CO₂e (kg ECM)⁻¹) than the warm (1.07 kg CO₂e (kg ECM)⁻¹). The range of emission
213 intensities (direct + indirect) was 0.09 kg CO₂e (kg ECM)⁻¹, the highest being the cool
214 climate, sandy soil and grass only, and the lowest the warm climate, sandy soil and
215 grass + maize.

216

217 Table 2 here

218

219 *Production characteristics*

220 DairyWise predicted a significantly higher number of dairy cows could be maintained
221 than the other models (Fig. 1A). This was not due to lower values for the DM intake
222 necessary to achieve the prescribed production; cow DM intake was on average
223 16.5, 15.6, 17.6 and 16.0 kg day⁻¹ for DairyWise, FarmAC, HolosNor and SFARMOD
224 respectively and for the followers, 6.0, 5.7, 7.1 and 4.8 kg day⁻¹ respectively. The
225 average milk production values ranged from 10413 litres ha⁻¹ for DairyWise to 8750

226 litres ha⁻¹ for HolsNor. The variation between scenarios was greatest for FarmAC
227 (HolsNor used the same livestock numbers as FarmAC). There were significant
228 differences between models in the amounts of concentrate feed imported (Fig. 1B),
229 reflecting the differences in the diet predicted or considered necessary to achieve the
230 target milk production specified. There were also large differences between models
231 in the extent to which the feed import varied between scenarios. The area dedicated
232 to maize silage production on grass/maize farms was significantly lower for
233 SFARMMOD than for the other models (Fig. 1C). Note that for DairyWise, the area
234 would have been higher, had the model not included a cap of 20% of field area that
235 could be allocated to maize cultivation. There were significant differences between
236 models in the amounts of fertiliser N applied (Fig. 1D).

237

238 Fig 1 here

239

240 *Farm-scale GHG emissions and emissions intensity*

241 Total GHG emissions expressed on an area basis were highest in DairyWise (Fig.
242 2A), significantly so in relation to SFARMMOD. However, this mainly reflects the
243 significantly higher number of livestock predicted by DairyWise. When expressed in
244 terms of an emission intensity, the differences between models were reduced,
245 although there was a significant difference between FarmAC and both DairyWise and
246 SFARMMOD (Fig. 2B). The range of the mean and median emission intensities was
247 0.08 and 0.10 kg CO₂e (kg ECM)⁻¹ respectively. Across scenarios, the range of
248 emission intensities was greatest for DairyWise (0.16 kg CO₂e (kg ECM)⁻¹) and least
249 for HolsNor (0.06 kg CO₂e (kg ECM)⁻¹). To remove the consequences of the higher

250 livestock number predicted by DairyWise, the remaining emissions will be expressed
251 as emissions intensities rather than on an area basis.

252

253 Figure 2 here

254

255 *Direct and indirect greenhouse gas emissions*

256 The enteric CH₄ emissions simulated by SFARMMOD were significantly greater than
257 those by FarmAC and HolosNor (Fig. 3A). SFARMMOD estimates enteric CH₄
258 emissions from milk production, hence the lack of variation between scenarios. There
259 were no significant differences between the estimates of field N₂O emissions from the
260 different models (Fig. 3B). The manure CH₄ emissions estimated by SFARMMOD
261 were lower than those of the other models, significantly so in the case of FarmAC
262 (Fig. 3C). In contrast, for manure N₂O emissions (Fig. 3D), the emissions estimated
263 by HolosNor were higher than those of the other models, significantly so in the case
264 of DairyWise and SFARMMOD.

265

266 Figures 3 here

267

268 Indirect N₂O emissions resulting from NH₃ volatilisation and NO₃⁻ leaching (kg CO₂e
269 (kg ECM)⁻¹ are shown in Fig. 4. There were large and significant differences between
270 models for the N₂O emissions from both NH₃ volatilisation and NO₃⁻ leaching. The
271 emissions estimated by HolosNor were significantly higher than for one or several
272 models. For FarmAC, the emissions resulting from NO₃⁻ leaching were particularly
273 variable between scenarios. The variation in GHG emissions between models is
274 shown in Table 3. For each source, the mean of the emissions from the four models

275 is subtracted from the emission from the individual model. Note the emission
276 intensities are expressed in grams rather than kilograms CO₂e (kg ECM)⁻¹.

277

278 Figure 4 and Table 3 here

279

280 **Discussion**

281 *Effect of scenarios*

282 More concentrate feed was required to provide a balanced diet in the grass only
283 system than the grass/maize system (Table 3). This meant that the total amount of
284 feed available on the grass only farms was greater than for the grass/maize system,
285 so more cows could be carried. Less fertiliser is applied to the grass/maize system
286 than the grass only system, since the application of plant-available N specified for
287 maize was lower than that for grass. The enteric CH₄ emissions were lower for the
288 grass/maize system than the grass only, due to differences in diet. Manure CH₄
289 emissions were lower under the warm climate, due to the shorter housing period,
290 although this was partially offset by the higher temperature, which led to a higher CH₄
291 emission per tonne of manure produced. The lower manure N₂O emission in the
292 warm climate reflects the shorter housing season and consequent lower manure
293 production. In contrast to CH₄ emissions, none of the models varied N₂O emissions
294 according to temperature. The direct N₂O emissions were higher under the cool
295 climate, as more excreta passed through the manure management system, leading
296 to gaseous N emissions which lowered the concentration of plant-available N. The
297 total N applied was therefore greater than for the warm climate.

298 The N₂O emissions associated with NO₃⁻ leaching were greater for the sandy than
299 clayey soil, due to the lower ability of the former to retain water. The difference was

300 greatest for the warm climate, since the precipitation excess was greatest here. The
301 higher total GHG emissions for the grass only system than for the grass/maize
302 system reflect the higher contributions from a number of sources, but especially
303 enteric CH₄ emissions. The lower total GHG emissions in the warm climate
304 compared to the cold reflect the lower emissions associated with manure
305 management.

306 The total GHG emission intensities calculated here are similar to those found for
307 Western Europe by Gerber *et al.* (2013) (once pre- and post-farm emissions are
308 discounted), for Tasmania by Christie *et al.* (2011) and for Ireland by Casey and
309 Holden (2005) (at the area requirement found here of 0.92 and 0.95 m² (kg ECM)⁻¹
310 for the cool and warm climates respectively). In contrast, the values were lower than
311 the 1.2 kg CO₂e (kg ECM)⁻¹ found for Portuguese dairy farms by Pereira and
312 Trindade (2015) and higher than the 0.83 and 0.73 kg CO₂e (kg ECM)⁻¹ found by
313 O'Brien *et al.* (2011) when using the IPCC (2006) methodology with default and local
314 parameterisation respectively. The separate contributions of CH₄ and N₂O found here
315 (means of 0.67 and 0.26 kg CO₂e (kg ECM)⁻¹ respectively) were, however, higher
316 than those found by Gerber *et al.* (2011) (0.54 and 0.24 kg CO₂e (kg ECM)⁻¹
317 respectively, after adjusting to the GWP for CH₄ and N₂O of Myhre *et al.* (2013).

318

319 *Differences in production characteristics*

320 The scenario specifications defined key production characteristics and yet achieving
321 complete standardisation of farm management was not possible. The models differed
322 both in their description of biophysical responses/feedback mechanisms and in the
323 extent to which management functions were internalised. For example, when
324 estimating the livestock number that could be carried on the farm, the DairyWise

325 predictions were 15% higher than the other models (Fig. 1A). This occurred despite
326 the major drivers of production (DM intake, import of concentrate feed and available
327 N used for crop production) being similar or the same as the other models. To
328 achieve an appropriate feed ration on the grass only farms, all models predicted it
329 was necessary to import cereal feed. This import of feed increases the number of
330 livestock that can be carried on the farm. Since maize silage has a higher nutritional
331 value than grass, an appropriate feed ration could be more easily achieved from
332 within the farms' resources when maize silage was available on the farm.

333 Consequently, three of the four models found the need to import cereal-based feed
334 was lower for the grass/maize system than for the grass only system and hence
335 fewer livestock were carried (Fig. 1B); the exception being DairyWise. In DairyWise,
336 the maximum percentage of the area of maize silage (20%) permitted is embedded in
337 the model and corresponds to the derogation obtained by the Netherlands under the
338 EU Nitrates Directive (European Commission, 1991 and 2014), so a higher import of
339 concentrates is necessary to achieve an appropriate feed ration. Even the remaining
340 models show substantial differences in the area allocated to maize silage production
341 (Fig. 1C), reflecting the differences in the definition of an appropriate feed ration and
342 the maize silage production predicted per unit area. This highlights a major difference
343 between farm-scale models and those of individual farm components such as crops;
344 the latter are commonly driven by external management variables whereas these are
345 internalised to a varying extent within the farm-scale models.

346 Finally, the application of N fertiliser varied between models (Fig. 1D). Since the total
347 amount of plant-available N applied was prescribed here and were different for grass
348 and maize, the differences in the application of N fertilizer reflect the differences
349 between models in the estimation of the plant-availability of N in the animal manure,

350 and for grass/maize system, the relative areas allocated to grass and maize
351 cultivation. This in turn reflects differences in the N losses occurring in the manure
352 management system. The farm characterisation specified a higher input of plant-
353 available N to grassland than to maize, so differences between models in the areas
354 used to produce maize silage also lead to differences in the farm-scale demand for
355 fertiliser N.

356

357 *Differences in greenhouse gas emissions*

358 Average predicted total GHG emissions per farm were highest for DairyWise (Fig.
359 2A). Since milk yield per cow was prescribed, the differences in GHG emissions can
360 be accounted mainly by differences in the number of livestock that the models
361 predicted could be supported on the farms, hence the differences between models
362 decrease when emissions are expressed as emission intensities (Fig. 2B). The
363 variation in enteric CH₄ emissions (Fig. 3A) has complex origins. The models differed
364 in the methods used to determine the quantity and quality of feed appropriate to
365 achieve the specified milk production per cow. Since pasture quality is predicted by
366 DairyWise, the feed grass quality could not be standardised. This means there were
367 differences between models in the quantities and qualities of fresh grass, grass
368 silage and maize silage fed. Finally, there were differences in methods used to model
369 enteric CH₄ emissions, which varied from varying emission factors per feedstuff
370 (DairyWise), through the IPCC methodology (FarmAC, HolosNor), to a fixed factor
371 based on milk production (SFARMMOD). The differences between estimates of N₂O
372 emissions from the soil were not significant (Fig. 3B), but this was due to the
373 substantial variation between models in their response to the scenarios. All models
374 use algorithms similar to those used by IPCC (2006) and so are driven by the total

375 amount of N entering the soil. The input of plant-available N was prescribed here so
376 the total N input was largely decoupled from the behaviour of the livestock and
377 manure management modules. The estimates of the total N input to the soil differed
378 between models, since differences in the estimated loss of N in the manure
379 management system meant that they differed in their assessment of the plant-
380 availability of N in the manure ex storage. The lower the plant-availability in the
381 manure, the higher the total manure N input. Furthermore, the total plant-available N
382 application to grass was prescribed to be higher than that to maize, so differences
383 between models in the allocation of land to these two crops affected the farm scale
384 input of N to the soil for the grass/maize systems.

385 The differences in GHG emissions from manure (Fig. 3C and 3D) reflect differences
386 in the management (see Farm management) and the throughput of manure dry
387 matter (DM) and N, resulting from differences in the methods used to estimate DM
388 and N excretion. The significant differences in indirect GHG emissions associated
389 with NH_3 volatilisation (Fig. 4A) reflect differences in assumptions made or the
390 methodology used. In particular, in the DairyWise simulations, a high DM content of
391 the applied slurry was assumed, leading to high field NH_3 emissions. In the FarmAC
392 simulations, a lower DM content was assumed and in SFARMMOD, a constant factor
393 independent of DM. The low indirect emissions of N_2O associated with NO_3^- leaching
394 predicted by DairyWise (Fig. 4B) is because it simulated a large loss of N via
395 denitrification on the clayey soil. The small effect of soil type on the HolosNor
396 simulations were because this model uses a leaching fraction that is not sensitive to
397 soil type. In contrast, FarmAC was highly sensitive to soil type, especially in the warm
398 climate due to the greater precipitation excess (difference between precipitation and
399 evapotranspiration).

400

401 *Predicting GHG emission intensities*

402 The total emission intensities calculated by the different models were similar but this
403 disguised differences between estimates of all the contributory emissions (Table 3).
404 Nevertheless, all models indicated that enteric CH₄ was the major source, followed
405 by soil N₂O emissions, and that the two together contributed more than half the total
406 emissions. This would be expected from earlier investigations (FAO, 2010, Gerber *et al.*
407 *et al.*, 2011). Furthermore, all models ranked the importance of the remaining sources
408 in the same order; manure CH₄ > indirect emissions > manure N₂O. This is important,
409 since the ranking of targets for mitigation measures is a common reason for
410 constructing such models (Cullen and Eckard, 2011, Del Prado *et al.*, 2013, Eory *et al.*
411 *et al.*, 2014). However, there were often significant differences between models in the
412 estimated emission from a given source, as a result of differences in the relationships
413 used to estimate GHG emissions, their parameterisation or the production
414 characteristics driving those relationships.

415 Variation between scenarios might be expected to increase with model complexity,
416 since this should increase the capacity to reflect the effect of different management
417 strategies (Beukes *et al.*, 2011). Cullen and Eckard (2011) estimated GHG emissions
418 for 4 locations in Australia and found the emissions estimated using the complex,
419 dynamic model DairyMod (Johnson *et al.*, 2008) to be between +10% and -30% of
420 the values estimated by an inventory method, depending on location. The majority of
421 the variation between the two methods arose from differences between locations in
422 the direct and indirect N₂O emissions predicted by the complex model. In the current
423 study, the range of emission intensities, relative to the model returning the lowest
424 estimate, was 4-9% for the cold climate and 13-16% for the warm climate. The lower

425 variation found in this study is probably because the representation of the two
426 dominant emission processes (enteric CH₄ and soil N₂O emissions) was in all models
427 based to varying degrees on that of the IPCC (2006) methodology.
428 In O'Brien *et al.* (2011), the use of locally-determined rather than default parameters
429 for the IPCC (2006) methodology led to a reduction in estimated GHG emissions of
430 about 13%. In this study, the emission factors in FarmAC and HolosNor were
431 adjusted to the IPCC (2006) default values for the relevant climate whereas the
432 parameter values are not climate-sensitive in DairyWise and SFARMOD. Since the
433 latter two models were developed in The Netherlands and UK respectively, this may
434 explain the larger variation between the model emission estimates for the warm
435 climate.

436

437 **Conclusions**

438 The difference between the models with the lowest and highest GHG emission
439 intensities, averaged over the six scenarios (0.08 kg CO₂e (kg ECM)⁻¹), was similar to
440 the difference between the scenarios with the lowest and highest emission intensities
441 (0.09 kg CO₂e (kg ECM)⁻¹), averaged over the four models. Furthermore, the
442 differences in the emission intensities between model estimates for most individual
443 sources were proportionately larger than at the farm scale but without any consistent
444 ranking of the models. The first conclusion is that if benchmarking is to contribute to
445 the quality assurance of emission estimates, there needs to be further discussion
446 between modellers, and between modellers and those with expert knowledge of
447 individual emission sources, concerning the nature and detail of the algorithms
448 needed; a process that is similar to that undertaken for ammonia emission modelling
449 (www.eager.ch, Reidy *et al.*, 2008). This process is particularly relevant for those

450 agriculturally-intensive Member States facing ambitious reduction targets within the
451 ESR, since the potentially high costs of mitigation measures may justify more
452 detailed modelling of individual sources (e.g. as is the case in The Netherlands;
453 Bannink *et al.*, 2011). Even though key production characteristics were standardised
454 in the scenarios used here, there were still significant differences between models in
455 the milk production ha⁻¹ and the amounts of N fertiliser and concentrate feed
456 imported. This was because the models differed both in their description of
457 biophysical responses/feedback mechanisms and in the extent to which
458 management functions were internalised. The second conclusion is that
459 benchmarking farm models for ruminant livestock systems will be more difficult than
460 for other agricultural production systems, where feedback mechanisms are less
461 pronounced.

462

463 **Acknowledgements**

464 This paper was supported by the FACCE-JPI knowledge hub Modelling European
465 Agriculture with Climate Change for Food Security (MACSUR). National funding was
466 received from the Norwegian Research Council and BBSRC (BB/N00485X/1 and
467 BB/K010301/1 projects). Authors acknowledge the financial support from the
468 European Commission (AIR3-CT94-1584, ECFARM project), UK Government
469 Department for Agriculture and Rural Affairs (WA0801, MEASURES project), Ministry
470 of Agriculture Fisheries and Food/Biotechnology and Biological Sciences Research
471 Council who funded parts of the development of SFARMMOD. Thanks also to Jordi
472 Doltra for his insight into dairy farm management in Northern Spain, Arne Oddvar
473 Skjelvåg for preparing the soil and climate data for HolosNor simulations, and Bente

474 Aspeholen Åby, Sissel Hansen and Tonje Storlien for sharing their experiences with
475 HolosNor.

476

477 **References**

478

479

480 Annetts JE and Audsley E 2002. Multiple objective linear programming for environmental
481 farm planning. *Journal of the Operational Research Society* 53, 933-943.

482 Bannink A, van Schijndel MW and Dijkstra J 2011. A model of enteric fermentation in dairy
483 cows to estimate methane emission for the dutch national inventory report using the
484 IPCC Tier 3 approach. *Animal Feed Science and Technology* 166-67, 603-618.

485 Bellarby J, Tirado R, Leip A, Weiss F, Lesschen JP and Smith P 2013. Livestock greenhouse
486 gas emissions and mitigation potential in Europe. *Global Change Biology* 19, 3-18.

487 Beukes PC, Gregorini P and Romera AJ 2011. Estimating greenhouse gas emissions from
488 new zealand dairy systems using a mechanistic whole farm model and inventory
489 methodology. *Animal Feed Science and Technology* 166-67, 708-720.

490 Bonesmo H, Skjelvag AO, Janzen HH, Klakegg O and Tveito OE 2012. Greenhouse gas
491 emission intensities and economic efficiency in crop production: A systems analysis of
492 95 farms. *Agricultural Systems* 110, 142-151.

493 Brentrup F, Kusters J, Lammel J and Kuhlmann H 2000. Methods to estimate on-field
494 nitrogen emissions from crop production as an input to LCA studies in the agricultural
495 sector. *International Journal of Life Cycle Assessment* 5, 349-357.

496 Casey JW and Holden NM 2005. The relationship between greenhouse gas emissions and
497 the intensity of milk production in Ireland. *Journal of Environmental Quality* 34, 429-
498 436.

499 Christie KM, Rawnsley RP and Eckard RJ 2011. A whole farm systems analysis of
500 greenhouse gas emissions of 60 Tasmanian dairy farms. *Animal Feed Science and*
501 *Technology* 166-67, 653-662.

502 Crosson P, Shalloo L, O'Brien D, Lanigan GJ, Foley PA, Boland TM, Kenny DA, 2011. A
503 review of whole farm systems models of greenhouse gas emissions from beef and
504 dairy cattle production systems. *Animal Feed Science and Technology* 166-67, 29-45.

505 Cullen BR and Eckard RJ 2011. Impacts of future climate scenarios on the balance between
506 productivity and total greenhouse gas emissions from pasture based dairy systems in
507 south-eastern Australia. *Animal Feed Science and Technology* 166-67, 721-735.

508 Del Prado A, Crosson P, Olesen JE and Rotz CA 2013. Whole-farm models to quantify
509 greenhouse gas emissions and their potential use for linking climate change mitigation
510 and adaptation in temperate grassland ruminant-based farming systems. *Animal* 7,
511 373-385.

512 Eory V, MacLeod M, Shrestha S and Roberts D 2014. Linking an economic and a life-cycle
513 analysis biophysical model to support agricultural greenhouse gas mitigation policy.
514 *German Journal of Agricultural Economics* 63, 133-142.

515 Erbach G 2016. Effort sharing regulation, 2021-2030. Limiting member states' carbon
516 emissions. In European Parliamentary Research Service.
517 [http://www.europarl.europa.eu/RegData/etudes/BRIE/2016/589799/EPRS_BRI\(2016\)5](http://www.europarl.europa.eu/RegData/etudes/BRIE/2016/589799/EPRS_BRI(2016)589799_EN.pdf)
518 [89799_EN.pdf](http://www.europarl.europa.eu/RegData/etudes/BRIE/2016/589799/EPRS_BRI(2016)589799_EN.pdf)

519 European Commission 1991. Council directive of 12 december 1991 concerning the
520 protection of waters against pollution caused by nitrates from agricultural sources.
521 *Official Journal of the European Communities* No L 375/1.

522 European Commission 2009. Decision no 406/2009/ec of the European Parliament and of
523 the Council of 23 april 2009 on the effort of member states to reduce their greenhouse
524 gas emissions to meet the community's greenhouse gas emission reduction
525 commitments up to 2020. *Official Journal of the European Union* L 140/136 - 148.

526 European Commission 2014. Commission implementing decision of 16 may 2014 granting a
527 derogation requested by the Netherlands pursuant to council directive 91/676/eec
528 concerning the protection of waters against pollution caused by nitrates from

529 agricultural sources (notified under document c(2014) 3103). [http://eur-](http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32014D0291)

530 [lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32014D0291](http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32014D0291)

531 FAO 2010. Greenhouse gas emissions from the dairy sector. A life cycle assessment. UN

532 Food and Agriculture Organisation, Rome, Italy.

533 Gerber P, Vellinga T, Opio C and Steinfeld H 2011. Productivity gains and greenhouse gas

534 emissions intensity in dairy systems. *Livestock Science* 139, 100-108.

535 Gerber PJ, Steinfeld H, Henderson B, Mottet A, Opio C, Dijkman J, Faluccci A and Tempio G

536 2013. Tackling climate change through livestock – a global assessment of emissions

537 and mitigation opportunities. UN Food and Agriculture Organisation, Rome, Italy.

538 Hagemann M, Ndambi A, Hemme T and Latacz-Lohmann U 2012. Contribution of milk

539 production to global greenhouse gas emissions. *Environmental Science and Pollution*

540 *Research* 19, 390-402.

541 IPCC 2006. 2006 IPCC guidelines for national greenhouse gas inventories, prepared by

542 the national greenhouse gas inventories programme (eds. S Eggleston, L Buendia, K

543 Miwa, T Nagara and K Tanabe), Japan.

544 Jarvis SC, Hutchings NJ, Brentrup F, Olesen JE and van der Hoek K 2011. Nitrogen flows in

545 farming systems across europe. In *European nitrogen assessment* (eds. MA Sutton,

546 CM Howard, JW Erisman, G Billen, A Bleeker, P Grennfelt, Hv Grinsven and B

547 Grizzetti), pp. 211-218, Cambridge University Press.

548 Johnson IR, Chapman DF, Snow VO, Eckard RJ, Parsons AJ, Lambert MG and Cullen BR

549 2008. Dairymod and ecomod: Biophysical pasture-simulation models for australia and

550 new zealand. *Australian Journal of Experimental Agriculture* 48, 621-631.

551 Kipling RP, Bannink A, Bellocchi G, Dalgaard T, Fox NJ, Hutchings NJ, Kjeldsen C, Lacetera

552 N, Sinabell F, Topp CFE, van Oijen M, Virkajarvi P and Scollan ND 2016. Modeling

553 European ruminant production systems: Facing the challenges of climate change.

554 *Agricultural Systems* 147, 24-37.

555 McGinn SM 2006. Measuring greenhouse gas emissions from point sources in agriculture.

556 *Canadian Journal of Soil Science* 86, 355-371.

557 Myhre G, D. , Shindell FM, Bréon W, Collins J, Fuglestedt J, Huang D, Koch JF, Lamarque
558 DL, Mendoza B, Nakajima T, Robock A, Stephens G, T. T and Zhang H 2013.
559 Anthropogenic and natural radiative forcing. In Climate change 2013: The physical
560 science basis. Contribution of working group I to the fifth assessment report of the
561 intergovernmental panel on climate change (eds. TF Stocker, D Qin, GK Plattner, M
562 Tignor, SK Allen, J Boschung, A Nauels, Y Xia, B V. and PM Midgley) Cambridge
563 University Press, Cambridge, UK.

564 Nemenyi PB 1963. Distribution-free multiple comparisons. PhD thesis, Princeton University,
565 USA.

566 O'Brien D, Shalloo L, Buckley F, Horan B, Grainger C and Wallace M 2011. The effect of
567 methodology on estimates of greenhouse gas emissions from grass-based dairy
568 systems. Agriculture Ecosystems & Environment 141, 39-48.

569 Özkan, Ş., Hill, J., 2015. Implementing innovative farm management practices on dairy
570 farms: a review of feeding systems. Turk. J. Vet. Anim. Sci 39, 1–9.

571 Pereira J and Trindade H 2015. Impact of the intensity of milk production on ammonia and
572 greenhouse gas emissions in Portuguese cattle farms. Spanish Journal of Agricultural
573 Research 13.

574 Tesco PLC 2016. Our carbon footprint. [https://www.tescopl.com/tesco-and-](https://www.tescopl.com/tesco-and-society/sourcing-great-products/reducing-our-impact-on-the-environment/our-carbon-footprint/)
575 [society/sourcing-great-products/reducing-our-impact-on-the-environment/our-carbon-](https://www.tescopl.com/tesco-and-society/sourcing-great-products/reducing-our-impact-on-the-environment/our-carbon-footprint/)
576 [footprint/](https://www.tescopl.com/tesco-and-society/sourcing-great-products/reducing-our-impact-on-the-environment/our-carbon-footprint/). Access date: 14 February 2017

577 Pohlert T 2014. The pairwise multiple comparison of mean ranks package (PMCMR) R
578 package. <http://CRAN.R-project.org/package=PMCMR>.

579 Reidy B, Dammgen U, Dohler H, Eurich-Menden B, van Evert FK, Hutchings NJ, Luesink
580 HH, Menzi H, Misselbrook TH, Monteny GJ and Webb J 2008. Comparison of models
581 used for national agricultural ammonia emission inventories in Europe: Liquid manure
582 systems. Atmospheric Environment 42, 3452-3464.

583 Schils RLM, de Haan MHA, Hemmer JGA, van den Pol-van Dasselaar A, De Boer JA, Evers
584 AG, Holshof G, van Middelkoop JC and Zom RLG 2007. Dairywise, a whole-farm dairy
585 model. Journal of Dairy Science 90, 5334-5346.

586 Vellinga TV, de Haan MHA, Schils RLM, Evers A and van den Pol-van Dasselaar A 2011.
587 Implementation of GHG mitigation on intensive dairy farms: Farmers' preferences and
588 variation in cost effectiveness. Livestock Science 137, 185-195.

589

590 **Table 1. Standardised farm data**

Category	Notes
Dairy cows	Mature live weight 600 kg, milk yield 7000 kg ECM cow ⁻¹ year ⁻¹ , diet: grass + concentrate or grass + maize silage + concentrate, grazing time: 16 hours day ⁻¹ during growing season*
Young animals	1 female:dairy cow, with male calves exported at birth, diet: grass + concentrate or grass + maize silage + concentrate, grazing time; 24 hours day ⁻¹ during growing season
Manure management	Livestock housing; freely-ventilated, fully slatted floor, manure storage; slurry tank with natural crust, manure application; broadcast spreader, no incorporation
Fields	Total area; 50 ha, irrigation; none
Crop potential DM yield (with irrigation if necessary)	Grass; cool climate: 10 tonnes ha ⁻¹ year ⁻¹ , warm climate: 8 tonnes ha ⁻¹ year ⁻¹ . Maize; cool climate: 14 tonnes ha ⁻¹ year ⁻¹ , warm climate: 18 tonnes ha ⁻¹ year ⁻¹ . Values were established after consultation with local experts.
N fertilisation	Grass; 275 kg plant-available N ha ⁻¹ year ⁻¹ . Maize 150 kg plant-available N ha ⁻¹ year ⁻¹ **

591 * cool climate; May to September, warm climate; March to November

592 ** Fertiliser type urea, with all fertiliser N considered plant-available. For animal manure,
593 plant-available N was equal to the mineral N present. The total N application in manure was
594 not permitted to exceed 250 kg N ha⁻¹ year⁻¹ for permanent grassland and 170 kg N ha⁻¹ year⁻¹
595 ¹ for maize silage. Manure was only exported if these application rates would otherwise be
596 exceeded.

597
598

Table 2 Summary of results for the different scenarios

	Scenario*							
	CSG	CSM	CCG	CCM	WSG	WSM	WCG	WCM
Number of dairy cows	69	62	69	63	70	65	69	67
Imported concentrate feed	126	67	124	82	116	67	116	78
Maize area	0	13	0	12	0	11	0	10
Fertiliser N	231	221	232	228	252	238	253	240
	kg CO ₂ e (kg ECM) ⁻¹							
	Direct emissions							
Enteric CH ₄	0.68	0.67	0.68	0.67	0.67	0.66	0.67	0.66
Manure CH ₄	0.14	0.14	0.14	0.14	0.11	0.11	0.12	0.11
Manure N ₂ O	0.03	0.02	0.03	0.02	0.02	0.02	0.02	0.02
Field N ₂ O	0.27	0.25	0.26	0.24	0.18	0.17	0.18	0.17
	Indirect emissions							
Volatilization of NH ₃	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.02
Leaching of NO ₃ ⁻	0.03	0.03	0.02	0.02	0.03	0.03	0.02	0.02
	Total emissions							
Emissions intensity	1.17	1.14	1.16	1.14	1.12	1.08	1.12	1.08

599 * Cxx = Cool climate, Wxx = Warm climate, xSx = Sandy soil, xCx = Clayey soil, xxG = Grass only,
600 xxM = Grass and maize.

601

602 **Table 3. Variation between models in the direct and indirect GHG emissions.**

Model	Enteric CH ₄	Soil N ₂ O	Manure CH ₄	Manure N ₂ O	Indirect	Direct + indirect
gCO ₂ e (kg ECM) ⁻¹						
DairyWise	0	-42	13	-7	0	-36
FarmAC	-23	33	48	0	-13	44
HolosNor	-8	-16	2	10	31	19
SFARMMOD	31	26	-63	-3	-17	-27
Mean of models	670	260	130	20	50	1130

603
604

605 **Figure captions**

606

607 **Figure 1**

608 The number of dairy cows (A), amount of concentrate feed imported (Mg DM year⁻¹)
609 (B), area of maize on farms growing both grass and maize (ha) (C) and fertiliser N
610 applied (kg ha⁻¹ year⁻¹) (D). The boxplots show the data median and quartiles.

611 Differences between models are not significantly different from one another if they
612 share the same letter.

613

614 **Figure 2**

615 Total GHG emissions from all sources, expressed as a farm total (kg CO₂e year⁻¹) (A)
616 and as an emission intensity (kg CO₂e (kg ECM)⁻¹) (B). The boxplots show the data
617 median and quartiles. Differences between models are not significantly different from
618 one another if they share the same letter.

619

620 **Figure 3**

621 Direct GHG emissions; enteric CH₄ emissions (A), soil N₂O emissions (B), manure
622 CH₄ (C) and manure N₂O emissions (D) (kg CO₂e (kg ECM)⁻¹). The boxplots show
623 the data median and quartiles. Differences between models are not significantly
624 different from one another if they share the same letter.

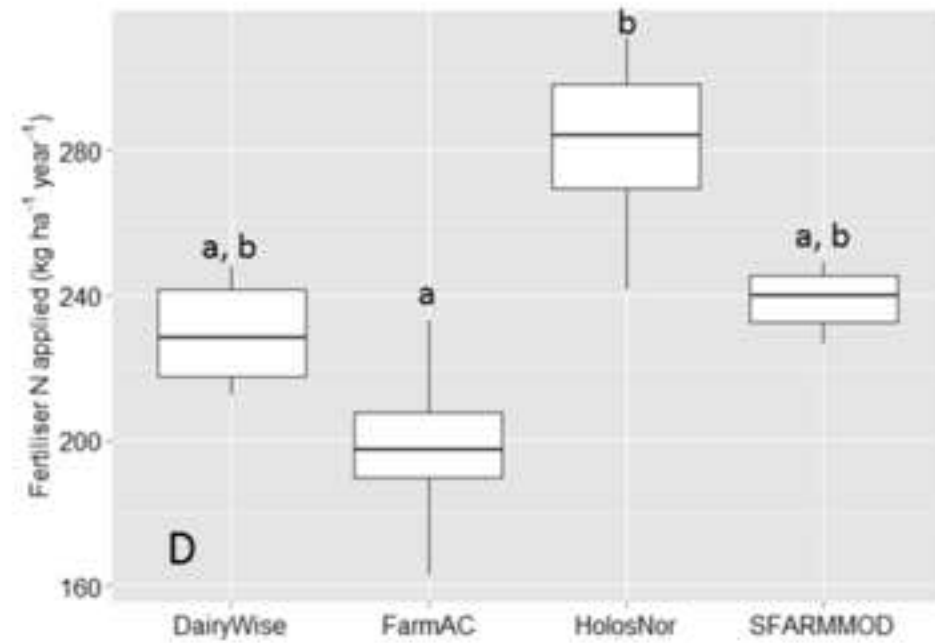
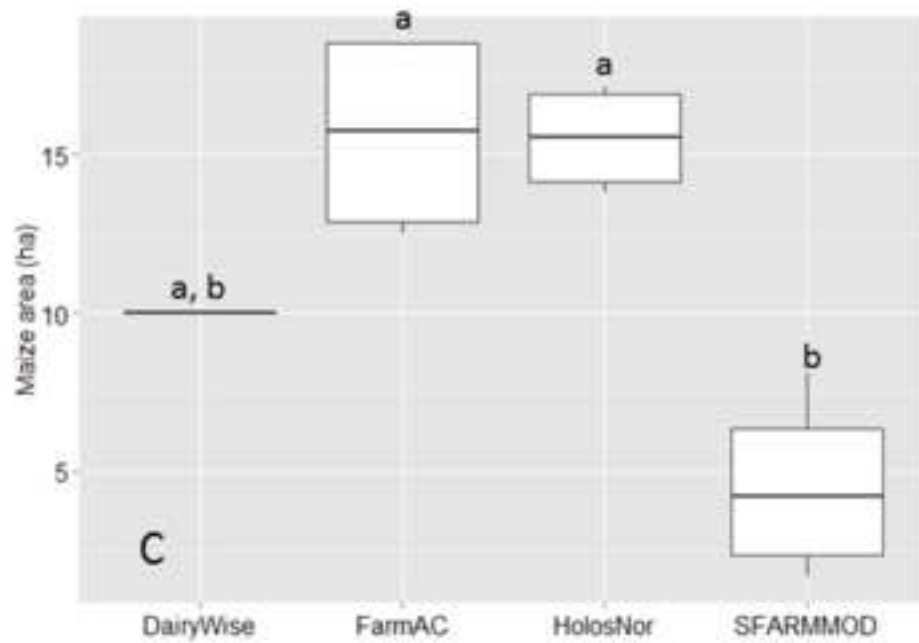
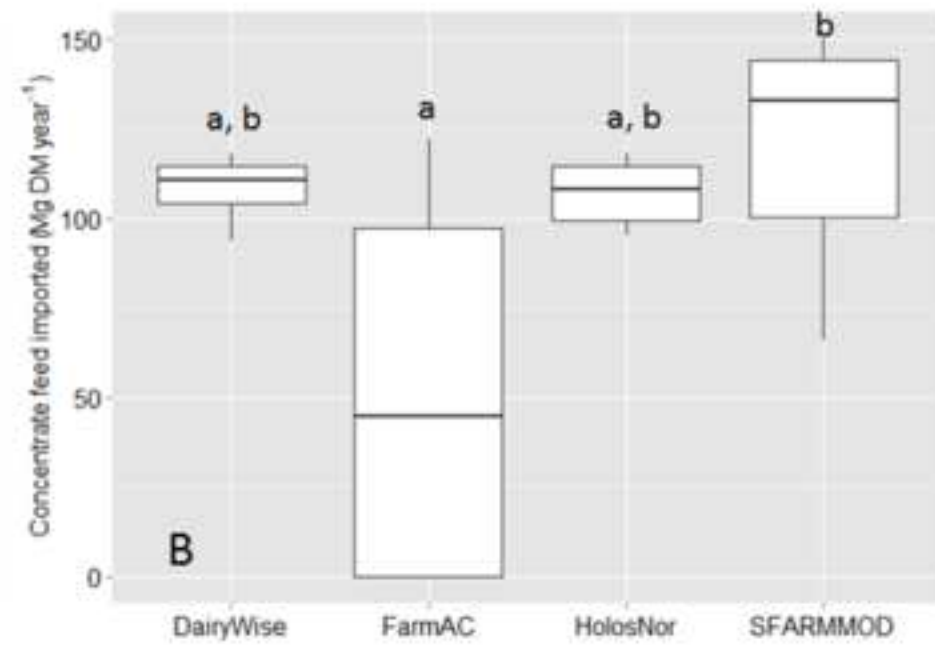
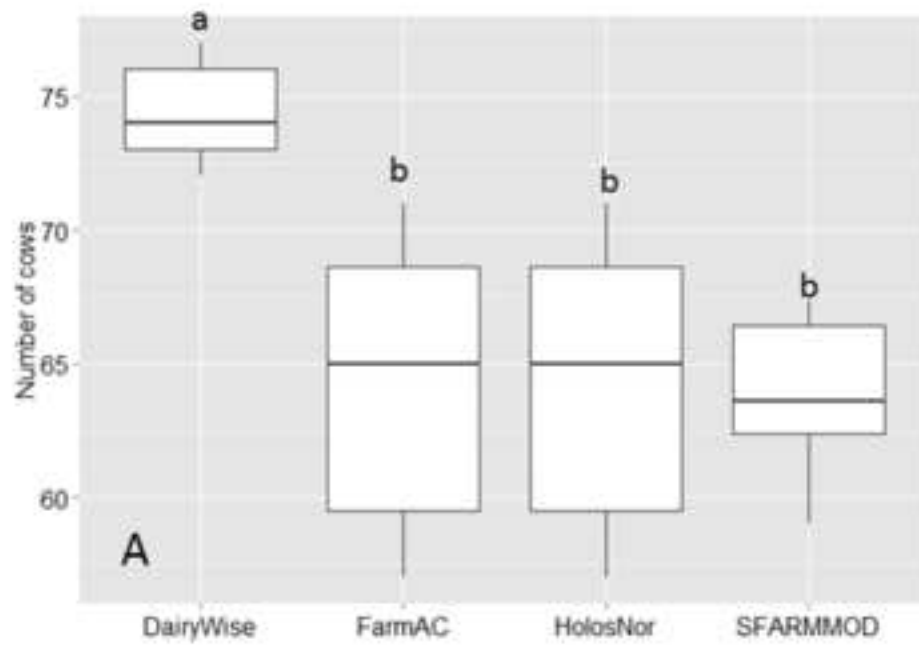
625

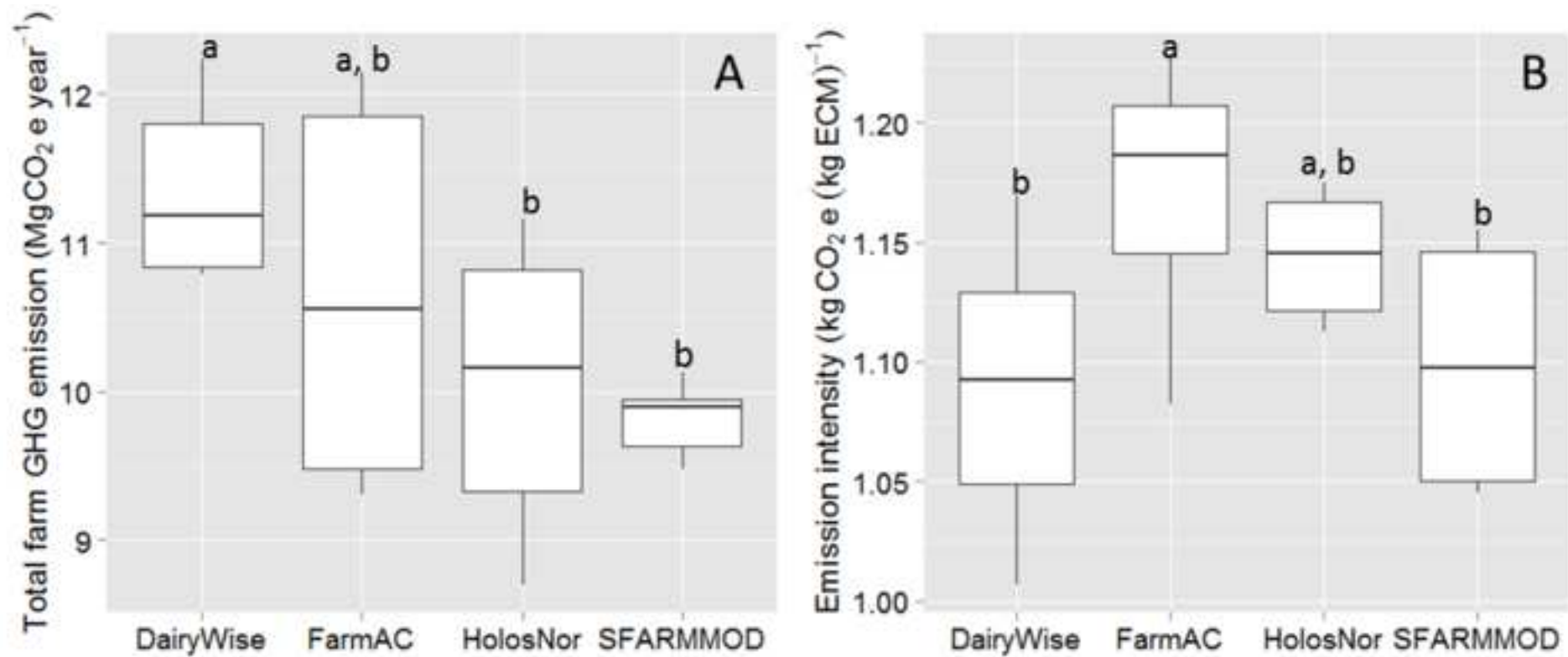
626 **Figure 4**

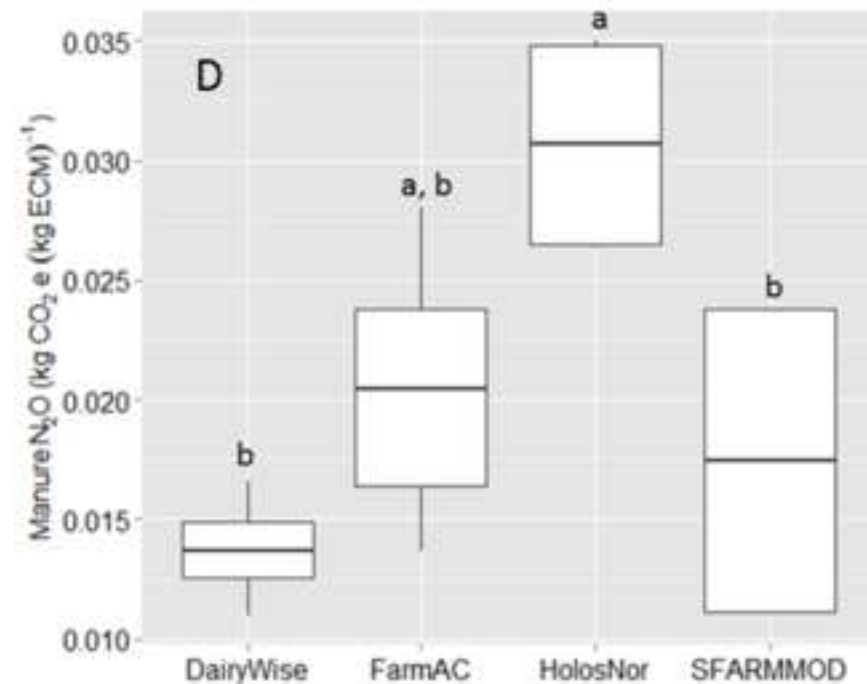
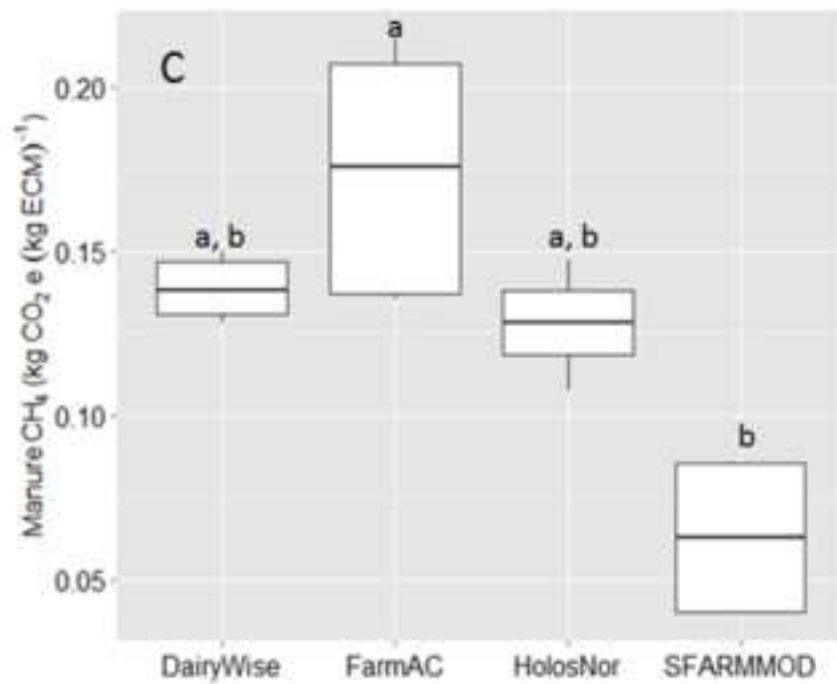
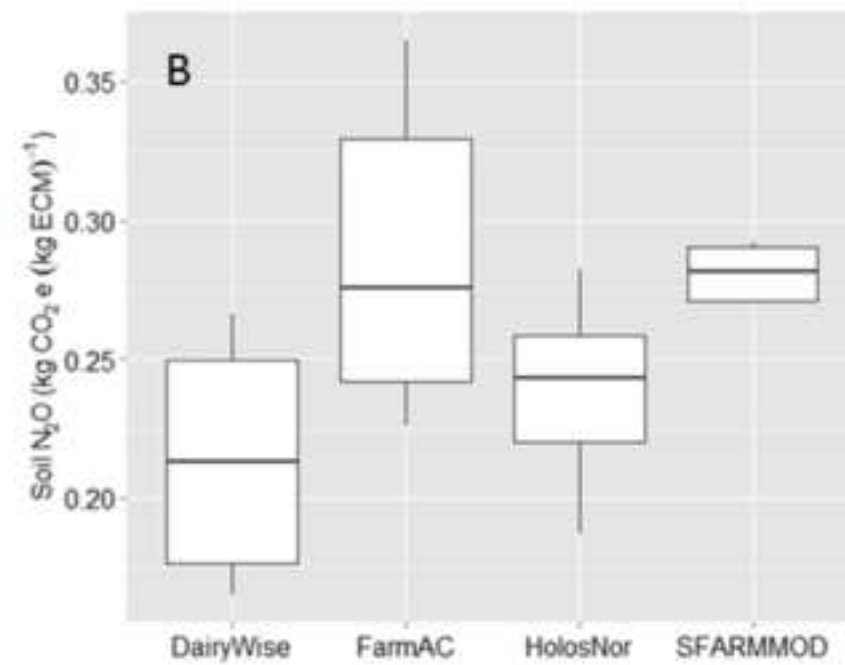
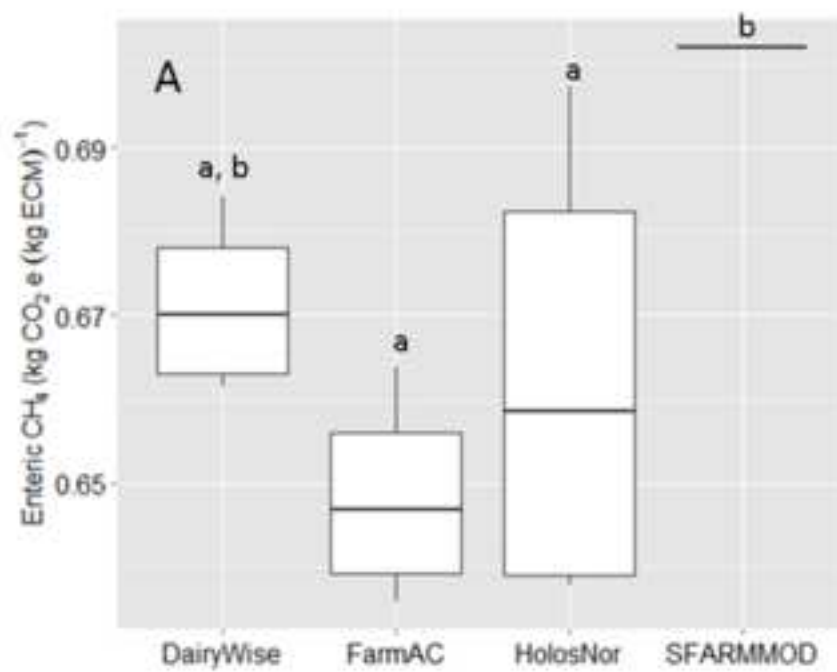
627

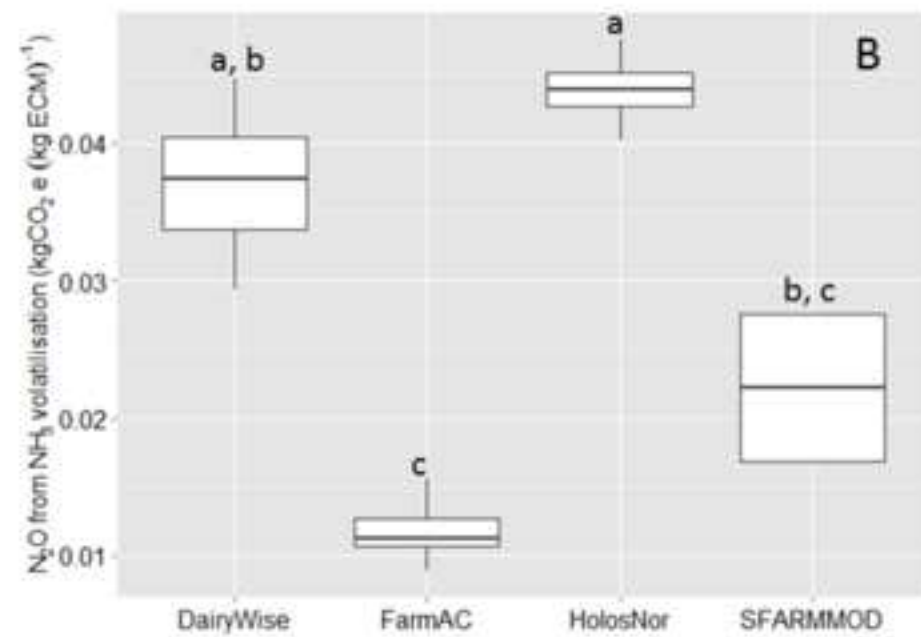
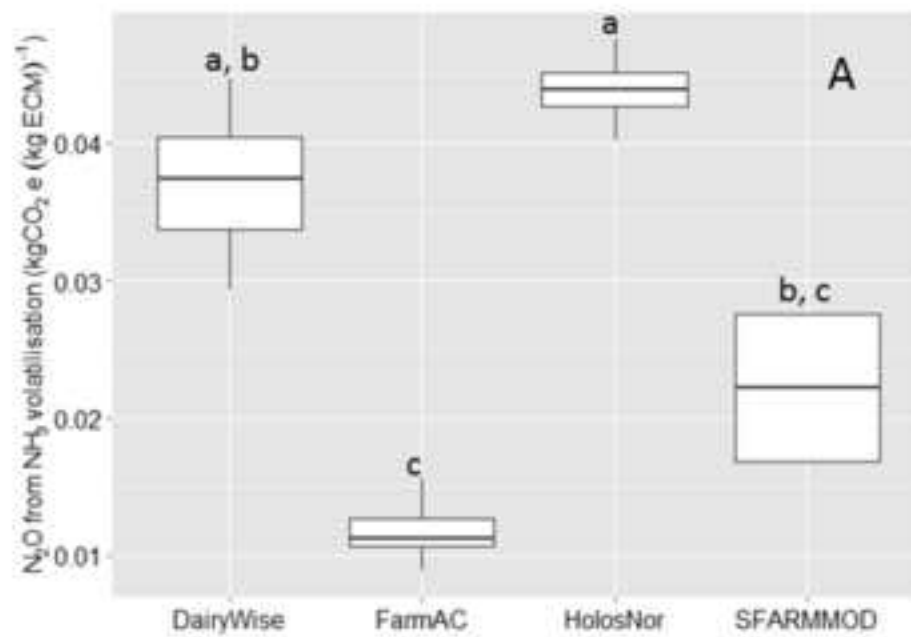
628 Indirect N₂O emissions resulting from leaching of NO₃⁻ (A) and from volatilisation of
629 NH₃ from manure management and field-applied manure (B) (kg CO₂e (kg ECM)⁻¹).

630 The boxplots show the data median and quartiles. Differences between models are
631 not significantly different from one another if they share the same letter.









1 **How do farm models compare when estimating greenhouse gas emissions from** 2 **dairy cattle production?**

3 N.J.Hutchings, S. Özkan Gülzari, M. de Haan and D. Sandars

4

5 **Models used**

6 *DairyWise*

7 The DairyWise model includes all major subsystems of a dairy farm. The central
8 component of DairyWise is the FeedSupply model, which meets the herd requirements for
9 energy and protein, using home-grown feeds (grazed or cut grass, forage crops e.g.
10 maize), maize silage and imported feed. The deficit between requirements and supply is
11 imported as concentrates and roughage (Alem and Van Scheppingen, 1993, Schroder *et*
12 *al.*, 1998, Zom *et al.*, 2002, Vellinga *et al.*, 2004, Vellinga, 2006, Schils *et al.*, 2007).
13 Methane, N₂O, and CO₂ emissions are calculated in the sub model GHG emissions, which
14 uses the emission factors from the Dutch emission inventories (Schils *et al.*, 2006).
15 Methane emissions from enteric fermentation are calculated using different emission
16 factors for concentrate, grass products, and maize (*Zea mays* L.) silage. The emission
17 factors used to calculate CH₄ emissions from manure storage are those used in the
18 MITERRA model (Velthof *et al.*, 2007), specific Dutch National Inventory Report
19 calculations, according to IPCC. Direct N₂O emissions are related to manure
20 management, N excreted during grazing, manure application, fertilizer use, crop residues,
21 N mineralization from peat soils, grassland renewal, and biological N fixation. The
22 emission factors are specified according to soil type and ground water level, with generally
23 higher emissions on organic soils and wetter soils. Indirect N₂O emissions resulting from
24 the partial denitrification of NO₃⁻ resulting from the oxidation of reduced N forms are

25 calculated based on NH₃ volatilization and NO₃⁻ leaching. The emissions of NH₃ volatilised
26 are calculated separately for animal housing, manure storage and field-applied manure
27 and fertiliser. Nitrate leaching to ground water was calculated for sandy soils according to
28 the NO₃⁻ leaching model of (Vellinga *et al.*, 2001). The amount of NO₃⁻ leached was related
29 to the amount of soil mineral nitrogen (SMN) to a depth of 1 meter at the end of the
30 growing season and soil type. The ground water table determined the partitioning of SMN
31 in NO₃⁻ leaching and denitrification. The lower the groundwater table, the higher the
32 proportion of NO₃⁻ leaching. For grassland, a basic SMN was calculated from the
33 difference between applied and harvested N. In the case of grazing, additional SMN was
34 calculated from urine excretions.

35

36 *FarmAC*

37 The FarmAC model simulates the flow of carbon (C) and N on arable and livestock farms,
38 enabling the quantification of GHG emissions, N losses to the environment and C
39 sequestration in the soil. It was constructed as part of the EU project AnimalChange
40 (<http://www.animalchange.eu/>). It is intended to be applicable to a wide range of farming
41 systems across the globe. The model is parameterised separately for each agro-climatic
42 zone.

43 A static livestock model is used in which the user defines the average annual number of
44 dairy cows, heifers and calves on the farm and the feed ration (including grazed forage).
45 Ruminant livestock production is modelled using a simplified version of the factorial energy
46 accounting system described in (CSIRO, 2007). Protein supply limitations on production
47 are simulated using an animal N balance approach. Losses of C in CO₂ and CH₄ are
48 simulated using apparent feed digestibility and IPCC (2006) Tier 2 methods, respectively.

49 Carbon and N in excreta are partitioned to grazed pasture in the same proportion as
50 grazed DM contributes to total DM intake, with the remainder partitioned to the animal
51 housing. Tier 2 methodologies are used for simulating flows in animal housing (CO₂ and
52 NH₃), manure storage (CO₂, CH₄, N₂O, N₂ and NH₃) and for N₂O, N₂ and NH₃ emissions
53 from fields. A dynamic model is used to simulate crop production and nutrient flows in the
54 field. The dynamics of soil C are described using the C-Tool model (Taghizadeh-Toosi et
55 al., 2014). A simple soil water model (Olesen and Heidmann, 1990) is used to simulate soil
56 moisture content and drainage. Soil organic N degradation follows C degradation. Mineral
57 N is not chemically speciated. The pool of mineral N is increased by the net mineralisation
58 of organic N and by inputs of fertiliser and manure. It is depleted by leaching, denitrification
59 and crop uptake. The N₂O emission associated with the modelled NH₃ volatilisation and
60 NO₃⁻ leaching were calculated using (IPCC, 2006). Crop production is determined by a
61 potential production rate, moderated by N and water availability. The user determines the
62 type, amount and timing of fertiliser and manure applications to each crop.

63

64 *HolosNor*

65 HolosNor was developed as a farm-scale model to calculate the GHG emissions produced
66 from combined dairy and beef production systems (Bonesmo *et al.*, 2012) in Norway. It is
67 based on the Canadian Holos model (Little, 2008) utilising the IPCC methodology (IPCC,
68 2006) modified for Norwegian conditions. The GHGs accounted for in HolosNor are CH₄
69 emissions from enteric fermentation and manure, direct N₂O emissions from agricultural
70 soils, indirect N₂O emissions resulting from NO₃⁻ leached, N in run-off and NH₃ volatilised.
71 Both direct and indirect N₂O emissions include emissions from manure and synthetic
72 fertiliser applications in soils.

73 The calculations of all emissions are explained in (Bonesmo *et al.*, 2012) in details based
74 on Tier 2 approach. Here only the modification made to the model and input parameters to
75 run the model are described. The ration consisted of grazed grass, grass silage (maize
76 silage in the grass and maize system) grown on farm and concentrates. There was no
77 crop production on the farm. Therefore, concentrates consisting of barley and soybean
78 meal were purchased outside the farm. The CO_{2e} emissions associated with production
79 of purchased concentrates were calculated from the mix of barley and soya that could
80 provide the amount of energy and protein in the purchased concentrate (Bonesmo *et al.*,
81 2012). The amount of concentrates required was calculated using a regression model (B.
82 Aspeholen Åby, Ås, 2016, personal communication) based on concentrate intake and
83 forage requirement for different levels of milk production, as described in (Volden, 2013).
84 Total net energy requirement (NE; MJ cow⁻¹ day⁻¹) was calculated based on the IPCC
85 (2006) recommendations considering maintenance, activity, lactation and pregnancy
86 requirements. Total NE requirement was then converted to DM by taking into account the
87 energy density of the feeds used (6 and 6.5 MJ NE (kg DM)⁻¹ for grass and maize silages,
88 respectively) (<http://feedstuffs.norfor.info/>). Silage requirement per cow was then
89 calculated by multiplying the total DM requirement by the silage proportion in the ration. By
90 dividing the total farm silage requirement by the potential DM yield given as an input
91 parameter (but corrected for fresh weight and feeding losses), the area to grow silage was
92 computed. The remainder area was allocated for grazing. In the maize scenario, the above
93 and below ground N residue concentration, yield ratio, and above and below ground
94 residue rations were adjusted according to (Janzen *et al.*, 2003). Methane conversion
95 factor for the warm climate was also adjusted according to IPCC guidelines, as the default
96 values represented the cool climate (IPCC, 2006). In calculating the soil and weather data

97 as one of the required input data, a 45% clayey soil for the Netherlands was found to be
98 outside the normal variation, and therefore the clay content of 35% was applied (A. O.
99 Skjelvåg, Ås, 2016, personal communication).

100

101 *SFARMMOD*

102 The Silsoe whole-FARM MODel is a linear programme (LP) that maximises long-run farm
103 profit. The concept and structure of the arable farm model are described in (Audsley,
104 1981) with the mathematical structure fully described in (Annetts and Audsley, 2002). The
105 latter paper details the extensions to model mixed arable and livestock systems. The main
106 focus of the environmental burdens concerns the N cycle. Methane emissions were also
107 included, but only from animal agriculture. Sources of information include inventories (Pain
108 *et al.*, 1997, Sneath *et al.*, 1997, Chadwick *et al.*, 1999) and experimental data and
109 mechanistic models (Scholefield *et al.*, 1991, Bouwman, 1996, Smith *et al.*, 1996,
110 Chambers *et al.*, 1999, MAFF, 2000). Some could be used directly (e.g. indirect N₂O
111 emissions associated with NH₃ volatilisation from animal houses), but others required
112 considerable adaptation to meet the long-term needs of the LP framework (e.g. NO₃⁻
113 leaching) and to ensure that nutrient cycles are closed with no change in N storage in the
114 soil (Williams *et al.*, 2002, Sandars *et al.*, 2003, Williams *et al.*, 2003).

115

116 **Supplementary references**

- 117 Alem vG and Van Scheppingen A 1993. The development of a farm budgeting program for dairy
118 farm. In Proceedings XXV CIOSTA-CIGR V Congress, pp. 326-331.
- 119 Annetts JE and Audsley E 2002. Multiple objective linear programming for environmental farm
120 planning. *Journal of the Operational Research Society* 53, 933-943.
- 121 Audsley E 1981. An arable farm model to evaluate the commercial viability of new machines or

122 techniques. Journal of Agricultural Engineering Research 26, 135-149.

123 Bonesmo H, Skjelvag AO, Janzen HH, Klakegg O and Tveito OE 2012. Greenhouse gas emission
124 intensities and economic efficiency in crop production: A systems analysis of 95 farms.
125 Agricultural Systems 110, 142-151.

126 Bouwman AF 1996. Direct emission of nitrous oxide from agricultural soils. Nutrient Cycling in
127 Agroecosystems 46, 53-70.

128 Chadwick DR, Sneath RW, Phillips VR and Pain BF 1999. A UK inventory of nitrous oxide
129 emissions from farmed livestock. Atmospheric Environment 33, 3345-3354.

130 Chambers BJ, Lord EI, Nicholson FA and Smith KA 1999. Predicting nitrogen availability and
131 losses following application of organic manures to arable land: Manner. Soil Use and
132 Management 15, 137-143.

133 CSIRO 2007. Nutrient requirements of domesticated ruminants. CSIRO PUBLISHING,
134 Collingwood, Australia.

135 IPCC 2006 2006 IPCC guidelines for national greenhouse gas inventories, prepared by the
136 national greenhouse gas inventories programme IGES Japan

137 Janzen HH, Beauchemin KA, Bruinsma Y, Campbell CA, Desjardins RL, Ellert BH and Smith EG
138 2003. The fate of nitrogen in agroecosystems: An illustration using canadian estimates.
139 Nutrient Cycling in Agroecosystems 67, 85-102.

140 Little S 2008 HoloS, a tool to estimate and reduce greenhouse gases from farms: Methodology &
141 algorithms for version 1.1.X. Agri-Food Canada.
142 <http://publications.gc.ca/pub?id=9.691658&sl=0>

143 MAFF 2000 Fertiliser recommendations for agricultural and horticultural crops (RB209) London,
144 UK

145 Pain B, Misselbrook T, Jarvis S, Chambers B, Smith K, Webb J, Phillips V, Sneath R and
146 Demmers T 1997 Inventory of ammonia emission from UK agriculture London, UK

147 Sandars DL, Audsley E, Canete C, Cumby TR, Scotford IM and Williams AG 2003. Environmental
148 benefits of livestock manure management practices and technology by life cycle assessment.
149 Biosystems Engineering 84, 267-281.

150 Schils RLM, Verhagen A, Aarts HFM, Kuikman PJ and Sebek LBJ 2006. Effect of improved
151 nitrogen management on greenhouse gas emissions from intensive dairy systems in the
152 Netherlands. Global Change Biology 12, 382-391.

153 Schils RLM, de Haan MHA, Hemmer JGA, van den Pol-van Dasselaar A, De Boer JA, Evers AG,
154 Holshof G, van Middelkoop JC and Zom RLG 2007. Dairywise, a whole-farm dairy model.
155 Journal of Dairy Science 90, 5334-5346.

156 Scholefield D, Lockyer DR, Whitehead DC and Tyson KC 1991. A model to predict transformations
157 and losses of nitrogen in UK pastures grazed by beef-cattle. Plant and Soil 132, 165-177.

158 Schroder JJ, Neeteson JJ, Withagen JCM and Noij IGAM 1998. Effects of N application on
159 agronomic and environmental parameters in silage maize production on sandy soils. Field
160 Crops Research 58, 55-67.

161 Smith JU, Bradbury NJ and Addiscott TM 1996. Sundial: A PC-based system for simulating
162 nitrogen dynamics in arable land. Agronomy Journal 88, 38-43.

163 Sneath R, Chadwick D, Phillips V and Pain B 1997 A UK inventory of methane emissions from
164 farmed livestock. MAFF, London, UK

165 Vellinga TV 2006. Management and nitrogen utilisation of grassland on intensive dairy farms. PhD
166 Thesis, Wageningen, The Netherlands.

167 Vellinga TV, Van der Putten AHJ and Mooij M 2001. Grassland management and nitrate leaching,
168 a model approach. Netherlands Journal of Agricultural Science 49, 229-253.

169 Vellinga TV, Andre G, Schils RLM and Oenema O 2004. Operational nitrogen fertilizer
170 management in dairy farming systems: Identification of criteria and derivation of fertilizer
171 application rates. Grass and Forage Science 59, 364-377.

172 Velthof GL, Oudendag DA and Oenema O 2007 Development and application of the integrated
173 nitrogen model MITERRA-Europe. Alterra-rapport 1663.1.
174 <http://content.alterra.wur.nl/Webdocs/PDFFiles/Alterrarapporten/AlterraRapport1663.1.pdf>
175 Volden H 2013 Yield level in milk production – feed and area requirement, Bioforsk Fokus 8: 42-44.
176 (in Norwegian).
177 Williams AG, Sandars DL and Audsley E 2002 Framework to evaluate farm practices to meet
178 multiple environmental objectives London, UK. Final report for Defra for Project Number WA
179 0801, London, UK.
180 Williams AG, Sandars DL, Annetts JE, Audsley E, Goulding KW, Leech P and Day W 2003. A
181 framework to analyse the interaction of whole farm profits and environmental burdens. In
182 Proceedings of EFITA 2003 4th Conference of the European Federation for information
183 technology in Agriculture, Food and Environment, Debrecen, Hungary, pp. 492–498.
184 Zom R, Van Riel J, André G and Van Duinkerken G 2002 Predicted intake with the cow model.
185 Praktijkonderzoek Veehouderij, Lelystad, The Netherlands (in Dutch)
186
187