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1 # Nitrogen losses to the environment following food-based digestate and

2 # compost applications to agricultural land

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13 **Abstract**

14 The anaerobic digestion of food waste for energy recovery produces a nutrient-rich digestate which

15 is a valuable source of crop available nitrogen (N). As with any 'new' material being recycled to

16 agricultural land it is important to develop best management practices that maximise crop available

17 N supply, whilst minimising emissions to the environment. In this study, ammonia (NH_3) and nitrous

18 oxide (N_2O) emissions to air and nitrate (NO_3^-) leaching losses to water following digestate, compost

19 and livestock manure applications to agricultural land were measured at 3 sites in England and

20 Wales. Ammonia emissions were greater from applications of food-based digestate (*c.*40% of total N

21 applied) than from livestock slurry (*c.*30% of total N applied) due to its higher ammonium-N content

22 (mean 5.6 kg/t compared with 1-2 kg/t for slurry) and elevated pH (mean 8.3 compared with 7.7 for

23 slurry). Whilst bandspreading was effective at reducing NH_3 emissions from slurry compared with

24 surface broadcasting it was not found to be an effective mitigation option for food-based digestate

25 in this study. The majority of the NH_3 losses occurred within 6 hours of spreading highlighting the

26 importance of rapid soil incorporation as a method for reducing NH_3 emissions. Nitrous oxide losses

27 from food-based digestates were low, with emission factors all less than the IPCC default value of 1%

28 (mean 0.45 ± 0.15%). Overwinter NO_3^- leaching losses from food-based digestate were similar to

29 those from pig slurry, but much greater than from pig farmyard manure or compost. Both gaseous N

30 losses and NO_3^- leaching from green and green/food composts were low, indicating that in these

31 terms compost can be considered as an 'environmentally benign' material. These findings have been

32   used in the development of best practice guidelines which provide a framework for the responsible

33   use of digestates and composts in agriculture.

34

35   **Capsule**

36   Field measurements of nitrogen losses as ammonia, nitrous oxide and nitrate leaching have been

37   used to develop best practice guidance for food-based digestate and compost use in agriculture.

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40

41   1. Introduction.

42   The United Kingdom generates around 14 million tonnes of food waste each year, the highest rate in

43   the European Union, which in total generates nearly 90 million tonnes; the quantity produced by the

44   different member states depends on numerous factors such as cultural practices, climate, diet and

45   socio-economic conditions (EC, 2010). A large proportion of this waste is disposed of to landfill, with

46   the UK sending around 8 million tonnes of biodegradable municipal waste (including food waste) to

47   landfill every year (Defra, 2016). Redirecting this material away from landfill will significantly reduce

48   greenhouse gas (GHG) emissions, in particular methane (CH_4), which has a global warming potential

49   around 25-fold greater than carbon dioxide (CO_2), thereby contributing to GHG reduction targets. To

50   this end, the EU Landfill Directive states that by 2020 the amount of biodegradable municipal waste

51   disposed of in landfill sites must be reduced by 65%, compared with 1995 levels (EC, 1999).

52   As part of the UK's commitment to reduce GHG emissions and to meet EU renewable energy targets,

53   policies and strategies have been implemented (DECC/Defra, 2011) to increase the treatment of

54   food waste through anaerobic digestion (AD; Styles *et al*., 2016) which, as well as providing

55   renewable energy, generates a nutrient-rich digestate or 'biofertiliser'. The most recent estimates

56   suggest that by 2020 around 5 Mt of the 7 Mt of food waste currently sent to landfill could be

57   available for digestion annually in the UK (DECC/Defra, 2011). However, AD is not without its

58   problems. In a recent study of the UK biogas sector, Styles *et al*. (2016) found that whilst biogas

59   energy has a lower GHG intensity than fossil fuels, it can increase acidification and eutrophication

60   burdens. Life-cycle assessment (LCA) studies have highlighted the fact that the environmental

61   outcomes are very sensitive to factors such as feedstock type, fugitive emissions, biomethane use,

62   energy conversion efficiency and digestate management. Nevertheless, Styles *et al.* (2016)

63 concluded that to maximise the potential for GHG abatement, climate change policies should

64 encourage digestion of food waste whilst restricting digestion of crop inputs and wastes that could

65 be used as animal feed.

66 The AD sector in the UK has been developing rapidly, with a 34% increase in the total number of

67 operational AD plants between 2012 and 2013, and an increase of 51% in the tonnage of organic

68 material processed (WRAP, 2014). Over 2 million tonnes of digestate were applied to agricultural

69 land in the UK in 2013, supplying a total of 92 kt nitrogen (N). Although this is only half the total N

70 supplied by composts, *c.*80% of the total N within food-based digestate is readily available (WRAP,

71 2016) i.e. in the form of ammonium-N (NH_4-N). A regular survey of the organics processing industry

72 has been undertaken since the mid-1990s; in 2012 the survey indicated that there had been little

73 year-on-year change in the UK composting sector, with a total of 3.5 million tonnes (fresh weight)

74 produced in 2012, 68% of which was recycled to agricultural land, supplying 192 kt N.

75 When applying organic materials such as digestate and compost to land, it is essential that their

76 application, agricultural or otherwise, is not harmful to the environment (i.e. to soil, water and air

77 quality) or human health. The European Nitrogen Assessment (Sutton *et al.,* 2011) highlighted how

78 the overall environmental costs of all N losses in Europe (estimated at €70–€320 billion per year at

79 current rates) outweigh the direct economic benefits of N in agriculture, due largely to loss of air and

80 water quality. The land application of organic materials therefore needs to be carefully managed to

81 maximise their crop available nutrient value and minimise their impact on the wider environment.

82 Food-based digestate is of particular interest being a new, less well understood material which will

83 vary in its compositional characteristics depending on the type of food waste used to produce it (e.g.

84 vegetable-based feed stocks produce digestate with lower N concentrations than those produced

85 from mixed food wastes). It generally has a relatively high total N content of around 5 kg/m^3 (WRAP,

86 2016) compared with *c.*3 kg/m^3 for cattle slurry and *c.*4 kg/m^3 for pig slurry (Defra, 2010) and might

87 therefore be expected to have the potential for greater N losses to the environment than livestock

88 manures or manure-based digestates.

89 Ammonia (NH_3) emissions to air contribute to acid deposition and can cause eutrophication of

90 sensitive ecosystems; in addition, NH_3 reacts with acids in the atmosphere to form particulate

91 matter which may pose a threat to human health (Webb *et al*., 2004). The amount and rate of NH_3

92 release following land spreading depends on a range of organic material (e.g. pH, readily available N,

93 dry matter), spreading (e.g. application rate, method and timing), soil (e.g. moisture content) and

94 environmental (e.g. temperature, wind speed, rainfall) factors (Nicholson *et al*., 2013). There is an

95 extensive body of research in the UK (and elsewhere) on NH_3 emissions following land application of

96   livestock manures and slurries (see for example Pain *et al.*, 1989; Chambers *et al.*, 1997; Sommer *et*
97   *al.*, 1997; Huijsmans *et al.,* 2001; Misselbrook *et al.*, 2002; Webb *et al.*, 2004), which has been used
98   to populate the National Ammonia Emissions Inventory (Misselbrook *et al.*, 2015) and provide
99   guidance for farmers to minimise NH_3 emissions from manures in the UK (Defra, 2009) and
100   elsewhere. Research has also been undertaken using manure-based and crop-based digestates
101   applied to land (e.g. Rubaek *et al.*, 1996; Wulf *et al.*, 2002); however, very little information is
102   available for food-based digestates produced and applied under conditions pertinent to the UK. A
103   recent study by Tiwary *et al.* (2015) showed that surface applied food-based digestate applications
104   led to NH_3 losses of 35-65% of the total N applied in the week following application, with an
105   abatement of 85% achieved if the material was incorporated into the soil immediately following
106   application. However, this study was undertaken at a field site in India and used digestates that may
107   not be comparable with those currently produced in the UK. Composts produced from green wastes
108   such as grass clippings and hedge trimmings (green compost) or from a mixture of green and food
109   wastes (green/food compost) tend to be applied to agricultural soils for soil conditioning purposes
110   (WRAP, 2016), although they also contain valuable amounts of plant available nutrients. The low
111   readily available N content of composts (generally <5% of total N; Defra, 2010) would suggest that
112   NH_3 losses following land spreading are also likely to be low, although there is little evidence
113   currently available to support this assertion.

114   Nitrous oxide (N_2O) is a greenhouse gas with a global warming potential *c.*300-fold greater than
115   carbon dioxide (IPCC, 2006). The UK Greenhouse Gas Emissions Inventory (2014) estimated that
116   *c.*70% of N_2O produced in the UK comes from agriculture (Brown *et al.,* 2016), of which the majority
117   (75%) is emitted from soils following N applications/returns (e.g. manufactured fertiliser N, crop
118   residue incorporation organic materials and urine from grazing returns) to land. Around 17% of
119   agricultural N_2O is emitted indirectly from soils following re-deposition of emitted NH_3 and from
120   leached nitrate (NO_3^-) (Brown *et al.*, 2016). As with NH_3, there is little information available on N_2O
121   losses following food-based digestate and compost applications to agricultural land. The current
122   IPCC Tier 1 default emission factor (EF) for N_2O losses from animal manure, compost, sewage sludge
123   and other organic N additions (e.g. digestates) is 1% of the total N applied (IPCC, 2006). By way of
124   comparison, Tiwary *et al*. (2015) found that N_2O emissions from food-based digestate were 4-10% of
125   the total N applied, which is much higher than the default IPPC EF of 1%, although these
126   measurements were made in India under very different soil and climatic conditions from those in the
127   UK. Measurements in Scotland showed that cumulative N_2O emissions following green compost
128   applied at 35, 100 and 200 t/ha ranged from 0.32 to 4.54 kg N_2O-N/ha/yr, with the higher values
129   measured following the 200 t/ha application in the wet spring of 2008 (Ball *et al.*, 2014). Although

130   compost application rates were very high in this experiment (35-200 t/ha compared with a more
131   typical rate of 30 t/ha), the maximum N_2O EFs were still only around 1% of the total N applied.

132   There is still much uncertainty over the factors which control N_2O emissions from food-based
133   digestates following application to agricultural land. For example, Pezzolla *et al*. (2012) found that
134   applying food –based digestate to a UK grassland did not increase emissions compared to the
135   untreated control, although measurements were made during an exceptionally dry growing season.
136   In contrast, an incubation study showed much larger emissions from food-based digestate compared
137   to ammonium sulphate applied under high soil moisture conditions (Koester *et al*., 2011). A later
138   study under similar conditions found emissions were twice as high from cattle slurry than from food-
139   based digestate (Koester *et al*., 2015). Following a laboratory incubation study using food-based and
140   other digestates, Rigby & Smith (2013) concluded that "the significance and influence of the
141   interaction between soil type and digestate stability and physical properties on denitrification
142   processes in digestate-amended soils require urgent investigation."

143   Additionally, poorly-managed applications of digestate and compost to land have the potential to
144   impact on water quality as a result of NO_3^- and other pollutant losses in drainage water. Indeed the
145   1991 EU Nitrates Directive (91/676/EEC) aims to protect water quality across Europe by preventing
146   NO_3^- from agricultural sources from polluting ground and surface waters by promoting good farming
147   practices. Again, there is a substantial body of evidence on NO_3^- losses following manure application
148   to agricultural land (e.g. Chambers *et al*., 2000; Smith *et al*., 2002), but very few field studies
149   available for food-based digestates. Goberna *et al*. (2011) reported that digestate application to soil
150   doubled NO_3^- leaching compared with animal manure applied at an equivalent N rate (80 kg N/ha),
151   however this was a pot-based study using manure-based digestate. Another pot experiment using
152   manure-based digestate applied to grass showed that there was less potential for NO_3^- leaching
153   losses compared with undigested slurry or mineral fertiliser (Walsh *et al.*, 2012).

154   The study described in this paper was part of a wider programme of work to establish a robust
155   scientific evidence base on the nutrient supply properties of digestate and compost applications to
156   land in the UK (Bhogal *et al*., 2016). The objective of the work described in this paper was to
157   measure N losses following autumn and spring applications of food-based digestate, green/food
158   compost and green compost to agricultural land via NH_3 volatilisation and N_2O emissions to air, and
159   NO_3^- leaching to water, and to compare these losses with those from livestock manures. The effect
160   of application techniques (i.e. surface broadcast compared with band spreading) were also assessed
161   to establish the effectiveness of this potential mitigation technique. The findings will be used to
162   develop best management practices that maximise crop nutrient utilisation, whilst minimising
163   environmental emissions of NH_3, N_2O and NO_3^-.

164   2. Materials and methods

165   2.1 Experimental sites

166   Experimental sites were established at 3 locations on a range of contrasting soil types and
167   agroclimatic zones (Table 1). To characterise each site, representative topsoil samples (0-15cm at the
168   arable site – Wensum - and 0-7.5cm at the grassland sites) were taken prior to the start of the
169   experiment and analysed for pH, sand (%), clay (%), silt (%), total N and organic carbon (C) using
170   standard methodologies (Anon., 1986), with the results shown in Table 1.

171   2.2 Treatments and design

172   The organic materials used were food-based digestate, compost, green or green/food compost,
173   green compost, solid farmyard manure (FYM) and livestock slurry. All organic materials were surface
174   broadcast, with additional treatments where food–based digestate and slurry were applied using a
175   bandspreading technique.

176   At WE, the only site where leaching losses to water were quantified, treatments were applied in
177   August 2011 (autumn), and then repeated in February 2012 (spring) on different experimental plots;
178   at the other sites, only spring applications (March 2012 at NW and May 2012 at PW) were evaluated.
179   For practical reasons, the spring and autumn treatments were grouped separately, with each group
180   of plots having an untreated control. Each treatment was replicated three times and arranged in a
181   randomised block design. Plot sizes were 3-7 m wide by 8-15m long.

182   Cattle FYM and slurries were used at NW and PW, and pig FYM and slurry at WE. The green compost
183   (PW and NW) or green/food compost (WE) and food-based digestates were sourced from
184   commercial enterprises local to the experimental sites, with the digestates from AD plants using
185   mainly commercial and municipal food wastes as a feedstock. The liquid materials (livestock slurry
186   and food-based digestate) were applied using a specially designed small plot applicator and the solid
187   materials (FYM and compost) were applied by hand (at the NW site the liquid organic materials were
188   also applied by hand). To be representative of commercial practice, at the grassland sites the
189   bandspread liquid material applications were by trailing shoe with 20cm spacing between the bands
190   and at WE, the bandspread applications were by trailing hose with 30cm spacing between the bands.
191   At WE, the autumn applied broadcast pig slurry and food-based digestate were incorporated into
192   the soil within 6 hours of application, using a rotavator to comply with Nitrate Vulnerable Zone (NVZ)
193   rules (SI, 2008; WSI, 2008).

194   The amount of organic material applied to each plot was weighed and recorded, so that the
195   application rate to each plot could be accurately determined, with the aim of applying the materials

196   at typical agronomic rates i.e. 20-30 m^3/ha digestate, 30-40 m^3/ha livestock slurry, 20 t/ha compost

197   and 25-30 t/ha FYM. Organic material application rates and N loadings are shown in Table 2.

198   2.3 Organic material analysis

199   At each site and application timing, a representative sample of each organic material type from each

200   experimental block was taken (*c.*2 litres or 2 kg per block), giving three replicate samples of each

201   material per site. These samples were analysed for pH, dry matter (DM), total N and NH_4-N using

202   standard methodologies (Anon., 1986).

203   2.4 Crop management

204   The grass at PW and NW was permanent pasture; the wheat crop at WE was grown according to

205   best farm practice using commercially recommended seed rates, with crop protection products

206   applied as needed and according to good agricultural practice to control weeds, pests and diseases

207   equally across all treatments. No manufactured N fertiliser was applied to the organic material

208   treatments, but all treatments, including the untreated control, had manufactured fertilisers (P, K

209   and S) applied based on the requirements of the untreated control (Defra, 2010).

210   2.5 Ammonia emissions

211   Wind tunnels were used to assess NH_3 emissions from the livestock manure treatments at each site,

212   based on the design developed by Lockyer (1984) and as recently used by Tiwary *et al*. (2015) and

213   Misselbrook *et al*. (2014). Each wind tunnel consisted of two parts; a transparent polycarbonate

214   canopy (2.0 m x 0.5 m) which covered the plot area, and a stainless steel duct housing a fan which

215   drew air through the canopy at a speed of 1 m/s; an anemometer measuring the wind speed, which

216   was recorded using a pulse counter. A sub-sample of the air entering and leaving the tunnel was

217   drawn through absorption flasks containing 80 ml of 0.02 M orthophosphoric acid. The absorption

218   flasks were changed after 24 hours and then daily for 7 days. The loss of NH_3 from beneath each

219   tunnel was calculated as the product of air flow through the tunnel and the difference between the

220   concentrations of NH_3 in the air entering (i.e. the background NH_3 concentration) and leaving the tunnel

221   as follows:

222   NH_3 loss = $((b/a)_{outlet} - (b/a)_{inlet} * c$ (1)

223   where, a is the volume of air sampled by each acid trap, b is the quantity of NH_3-N in each trap over

224   the sampling period and c is the volume of air drawn through each tunnel. The rate of loss was

225   calculated over each time period so that the pattern of loss could be quantified and cumulative

226   losses were then calculated by summing over all sampling periods.

227   2.6   Nitrous oxide emissions

228   Nitrous oxide emissions were measured using the static chamber technique (Chadwick *et al*., 2014),

229   from three replicate plots per treatment, using 5 chambers per plot (giving a total of 15 replicate

230   chambers per treatment). Each chamber had dimensions of 40 cm x 40 cm square and was 25 cm

231   tall, giving a soil surface area coverage of 0.16 m^2. The chambers were installed immediately after

232   organic material application and positioned in a 5 cm deep slot cut in the soil. The chambers were

233   designed to completely enclose growing arable crops and grassland, without damage, with chamber

234   extensions fitted to enable measurements to be taken from mature cereal and grass crops. On each

235   sampling occasion, the chambers were covered for at least 40 minutes before the headspace was

236   sampled. Sampling was normally conducted between 10 am and 2 pm, as previous studies have

237   shown that emissions at this time of day approximate to average diurnal emission rate (Clayton *et*

238   *al*., 1994). The samples were transferred to evacuated vials prior to Gas Chromatography analysis

239   using an Electron Capture Detector. To verify the assumption of linear gas accumulation within a

240   chamber's headspace, 3 chambers were selected on each sampling occasion from which a time

241   series of headspace samples was taken every 15 minutes up to 60 minutes after closure. The

242   following steps were taken to help ensure that linearity in gas accumulation was achieved, by (i)

243   ensuring an air-tight seal between the chamber and soil; (ii) ensuring an air-tight seal between the

244   chamber and lid; (iii) using 'large' chambers to provide as much headspace as practically possible,

245   whilst retaining analytical sensitivity.

246   Data from previous UK studies have indicated that *c.*75% of total *direct* N_2O emissions are likely to

247   occur in the first 4-6 weeks following slurry application. Therefore the sampling strategy was

248   weighted accordingly, with *c.*50% of sampling events carried out during the (likely) period of highest

249   N_2O fluxes (i.e. 4-6 weeks after application), giving a total of at least 30 measurements over a 12

250   month period*.*

251   2.7   N leaching losses

252   At WE, N leaching losses to water were measured from the plots receiving autumn applications of

253   organic materials, using Teflon cup water samplers. Five water samplers were installed on each plot

254   to a depth of 90 cm. Samples of soil water were collected every 2 weeks or after 50 mm drainage,

255   whichever occurred sooner, throughout the drainage period and analysed for NO_3^- and NH_4-N using

256   standard methodologies (Anon, 1986). Drainage estimates were obtained using IRRIGUIDE (Bailey &

257   Spackman, 1996) and were combined with the pollutant concentration data to calculate N losses in

258   drainage water.

259

260   2.8   Other measurements

261 Soil samples for mineral nitrogen (SMN) determination were taken periodically (from 0-10 cm depth)

262 throughout the experimental period to quantify changes in soil mineral N following the organic

263 material applications that could influence gaseous N losses. Soil moisture content measurements

264 (per block) were also made on each N_2O sampling occasion, and converted to water filled pore space

265 (WFPS) using the following formula:

266   WFPS (%) = Soil moisture (% w/w) x BD/(1 - (BD/2.65))   (2)

267 Where BD is the bulk density of the studied soil (g/cm^3) and 2.65 g/cm^3 is the bulk density of mineral

268 soils.

269 Daily rainfall and mean air and soil temperature (at 5cm depth) data were measured at each site or

270 obtained from a nearby meteorological station.

271   2.9 Statistical analysis

272 At each experimental site, conventional analysis of variance (ANOVA) comparisons were undertaken

273 between the different treatments in terms of cumulative N_2O and NH_3 emissions to air, and NO_3^- and

274 NH_4-N losses to water, with comparison of P statistics (quoted in the text). A separate ANOVA was

275 carried out at each site, after which *post-hoc* testing was undertaken to evaluate which treatment

276 means were different from each other using a Duncan's multiple range test (using Genstat version

277 12; VSN International Ltd, 2010).

278 Additionally, the pooled data were analysed to assess effects of N_2O and NH_3 emissions to air (*i.e.* a

279 cross-site analysis ANOVA was undertaken). This enabled us to assess whether a particular

280 treatment had had a statistically significant effect at each site, and across *all* study sites. The ANOVA

281 also indicated if there were significant interactions between sites and treatments.

282   3. Results

283   3.1 Organic material analysis

284 The composition of the organic materials applied at each site and application timing is shown in

285 Table 2. As expected the food-based digestate had higher total N contents (mean 7.0 kg/t across all 4

286 site/timings) than the pig slurry (mean 2.8 kg/t at WE) or cattle slurry (mean 2.4 kg/t at NW and PW).

287 The NH_4-N contents of the food-based digestate and pig slurry (*c.*80% of total N) were higher than

288 the cattle slurry (mean 54% of total N), with the solid materials containing <10% of the total N in the

289 NH_4-N form.

290 3.2 Ammonia emissions

291 Ammonia emissions from the spring 2012 applications at NW were similar from the food-based

292 digestate and cattle slurry (23-31% total N applied), and both these treatments had higher emissions

293 than the green compost and cattle FYM (<5% total N applied; $P<0.05$). Perhaps surprisingly, there

294 were no differences ($P>0.05$) in NH_3 emissions between the bandspread and broadcast food-based

295 digestate and cattle slurry treatments.

296 At PW, NH_3 emissions following the spring 2012 organic material applications were higher from

297 food-based digestate (48-63% total N applied) than from cattle slurry (12-37% total N applied), with

298 the green compost and cattle FYM treatments having much lower emissions (<3% total N applied;

299 $P<0.05$). Bandspreading reduced NH_3 emissions from the food-based digestate by $c.25\%$ (although

300 this was not significant; $P>0.05$) and cattle slurry by 70% ($P<0.05$) compared with the respective

301 broadcast applications.

302 At WE, NH_3 emissions following the autumn 2011 organic material applications were again greater

303 from food-based digestate (56-60% total N applied) than from pig slurry ($c.40\%$ total N applied), with

304 the pig FYM and green/food compost having the lowest emissions (4-14% total N applied; $P<0.05$).

305 There were no significant differences in NH_3 emissions between the bandspread and broadcast (soil

306 incorporation within 24 hours) food-based digestate and pig slurry applications probably because

307 the bandspread digestate and slurry stayed on the soil surface due to the dry (hydrophobic) soil

308 conditions. NH_3 emissions were lower following the spring 2012 food-based digestate and pig slurry

309 applications than in autumn because the soil was moist leading to more rapid soil infiltration

310 compared with the dry, hydrophobic soil surface of the arable stubble in autumn which restricted

311 infiltration. Ammonia emissions were similar from the food-based digestate and pig slurry

312 treatments ($c.20\%$ total N applied), with the pig FYM and green/food compost again having lower

313 emissions ($c.2\%$ total N applied; $P<0.05$). There were no significant differences between the

314 bandspread and broadcast food-based digestate and pig slurry treatments, which was most probably

315 due to the bandspread applications not staying in a band (plus some temporary ponding on the soil

316 surface), followed by 'rapid' infiltration into the soil on both treatments.

317 The cross-site analysis of the total NH_3 losses from the autumn 2011 applications at WE, and the

318 spring 2012 organic material applications at NW, PW and WE is summarised in Table 3. Ammonia

319 emissions were similar on the broadcast and bandspread food-based digestate ($c.40\%$ of total N

320 applied) and were greater than those from the broadcast and bandspread livestock slurry, with FYM

321 and compost having the lowest emissions. There was no difference in emissions due to spreading

322 method for the food-based digestate, but broadcast slurry gave rise to greater emissions than

323 bandspread slurry. Total NH_3 losses from food-based digestate were of a similar magnitude to the

324 35-65% of total N applied reported by Tiwary et al. (2015).

325 The cross-site NH_3 emission curves (Figure 1) indicate that the majority of the NH_3 losses from food-

326 based digestate and livestock slurry occurred within 6 hours of spreading.

327 3.3 Nitrous oxide emissions

328 Nitrous oxide fluxes at NW in spring 2012 were generally low throughout the measurement period

329 at <20 g N_2O-N/ha/day on all treatments (Figure 2a). There was a small peak in emissions ($c.$30 g

330 N_2O-N/ha/day) at the end of May which corresponded with a drop of $c.$10% in the WFPS. Cumulative

331 N_2O losses (net of the control) were all low (<0.5 kg N_2O/ha) and there were no significant

332 differences in EFs between the different treatments (Figure 2b).

333 At PW in spring 2012, N_2O fluxes were generally higher than at NW. Emissions peaked (particularly

334 on the FYM treatment at $c.$110 g N_2O-N/ha/day) around 2 weeks after the organic materials were

335 applied in early May; after this, emissions on all the organic material treatments returned to

336 background levels ($c.$10 g N_2O-N/ha/day; Figure 3a). As at NW, cumulative N_2O losses (net of the

337 control) were all low (<0.5 kg N_2O/ha). Emissions from the green compost treatment were

338 significantly lower than from the cattle slurry and FYM treatments (P<0.05). Although bandspreading

339 significantly increased the EF from cattle slurry compared with surface broadcasting (P<0.05), there

340 was no difference in the EF between the broadcast and bandspread digestates. However, the

341 bandspread digestate had a significantly (P<0.05) lower EF than the bandspread cattle slurry (Figure

342 3b).

343 Nitrous oxide fluxes at WE in autumn peaked at $c.$100 g N_2O-N/ha/day, shortly after the organic

344 materials were applied in early August 2011; emissions on all the organic material treatments had

345 returned to background levels ($c.$10 g N_2O-N/ha/day) by the end of November 2011 (Figure 4a). Net

346 cumulative N_2O losses ranged from 0 kg N_2O/ha on the compost treatment to 1.2 kg/ha on

347 broadcast digestate treatment. Although the EF for green/food compost was very low, and

348 bandspreading slurry and digestate resulted in numerical reductions in the EF, none of the treatment

349 effects were significant (Figure 4b).

350 At WE in spring, N_2O fluxes peaked at $c.$70 g N_2O-N/ha/day in March about 1 month after the organic

351 materials were applied in late February 2012; emissions on all the organic material treatments had

352 returned to background levels ($c.$10 g N_2O-N/ha/day) by the end of April 2012 (Figure 5a). Net
353 cumulative N_2O losses ranged from <0 kg N_2O/ha on the compost treatment to 1.8 kg/ha on the
354 bandspread digestate treatment. Emissions from the green compost were significantly lower than
355 from the digestate treatments and the bandspread pig slurry (P<0.05). There was no effect of
356 bandspreading on the EF for digestate or pig slurry compared with surface broadcasting (Figure 5b).

357 Overall, there was no significant effect of food-based digestate/livestock slurry application method
358 (i.e. surface broadcast compared with bandspreading) on N_2O emissions (Table 3).

359 3.3.1 Leaching losses

360 Drainage volumes at WE over the winter of 2011-12 were low (92 mm) due to the lower than
361 average over-winter rainfall of $c.$200mm compared to the 25 year average of just over 300 mm.
362 Nitrate concentrations in the drainage water were $c.$50 mg/l on all treatments at the start of
363 drainage in November/December 2011; concentrations peaked in January/February 2012 and were
364 highest ($c.$150 mg/l) on the surface broadcast food-based digestate treatment (Supplementary
365 Information Figure S1a). The peak in leachate NO_3^- concentrations occurred somewhat later than the
366 maximum soil NO_3^- concentrations which were measured in September 2011, shortly after the
367 organic materials were applied (Supplementary Information Figure S1b). This was probably because
368 the NO_3^- present in the top 15cm of the soil took this time to move through the soil profile to 30cm,
369 which was the depth at which the leachate was sampled; additional NO_3^- would also be mineralised
370 over the autumn period from the organic N applied with the organic materials, contributing to NO_3^-
371 leaching losses.

372 Cumulative NO_3^- leaching losses following the food-based digestate and pig slurry treatments were
373 greater (P<0.05) than from the pig FYM and compost treatments, with no significant differences
374 between bandspread and broadcast food-based digestate treatments (Figure 6). Ammonium-N
375 concentrations in the drainage waters were very low on all treatments (<0.05 mg/l) and cumulative
376 leaching losses were <0.02 kg/ha (i.e. <0.01% of the total N applied).

377 The winter wheat crop (drilled in early October) was only expected to take up 5-10 kg N/ha during
378 the autumn growth phase which effectively ceases by the end of October (Nicholson et $al.$, 2013).
379 This is equivalent to <10% of the total N applied with the organic materials and hence would have
380 had little effect on the quantities of N leached over the winter period from the different organic
381 materials.

382   4. Discussion

383   4.1 Ammonia emissions

384   The higher NH_3 emissions from the food-based digestate than from livestock slurry ($P<0.05$) were

385   most probably due to the greater NH_4-N content of the food-based digestate (mean 5.6 kg/t)

386   compared with the livestock slurries (mean 2.2 kg/t for pig slurry and 1.3 kg/t for cattle slurry), Table

387   2. Additionally, the mean pH of the food-based digestate was 8.5 compared with 7.8 for pig slurry

388   and 7.6 for cattle slurry (Table 2). It is known that pH values greater than 8 are particularly conducive

389   to elevated NH_3 emissions from digestates (e.g. Hoeksma *et al.*, 2012) and digestion of livestock

390   slurry has been shown to increase pH with a concomitant increase in NH_3 volatilisation (Sommer *et*

391   *al.*, 2006). Acidification (i.e. decreasing the pH) has been adopted as the Best Available Technology

392   (BAT) for reducing NH_3 losses from livestock slurry in some European countries (Kai *et al.*, 2008),

393   however further research into the costs, practicalities and effectiveness of acidification of food-

394   based digestates as a method of controlling NH_3 emissions is still required.

395   Overall, bandspreading was effective at reducing NH_3 emissions from livestock slurry compared with

396   surface broadcasting as reported in many previous studies (e.g. Sommer *et al.*, 1997; Webb *et al.*,

397   2005), because the bandspreading technique reduces the surface area of slurry exposed to the

398   atmosphere from which NH_3 can be volatilised. However in this study bandspreading was not found

399   to be an effective mitigation option for food-based digestate. Bandspreading of liquid organic

400   materials (such as food-based digestate) is now a common practice, with the majority of contractor-

401   spread digestate applied using bandspreaders. In this study, the failure to observe a reduction in NH_3

402   emissions when bandspreading food-based digestate (in comparison with surface broadcast

403   applications) was most probably due to soil and/or organic material properties that meant that the

404   digestate did not rapidly infiltrate into the soil or did not stay in a narrow band on the soil surface.

405   Dry matter content is known to affect NH_3 emissions from cattle slurry, with emissions increasing as

406   slurry dry matter content increases (e.g. Sogaard *et al.*, 2002; Misselbrook *et al.*, 2004); it is likely

407   that this relationship will also hold when comparing food-based digestates with different dry matter

408   contents, although we do not know of any research data specific to digestates. The importance of

409   crop height as a factor affecting NH_3 emissions is not clear; previous UK/Danish research has shown

410   that the abatement efficiency of bandspread slurry applications increased with grass height and was

411   typically 60% (Thorman *et al*, 2008). In contrast, more recent Irish data (Lalor *et al.*, 2012) has not

412   supported a relationship between NH_3 emission reductions from bandspread slurry applications and

413   grass height. There is a requirement for further research on this topic to enable us to better assess

414   the importance of crop height in controlling NH_3 losses from food-based digestates. It is important to

415 bear in mind that bandspreading technologies provide numerous other advantages over broadcast
416 applications (e.g. more even digestate application and hence more accurate assessment of
417 application rates, the ability to apply from tramlines, reduced odour and crop damage, and a cleaner
418 sward) implying that it is still the best application method available.

419 In line with findings from previous studies using livestock slurry (Nicholson *et al.*, 2013), Figure 1
420 indicates that the majority of the NH_3 losses from food-based digestate occurred within 6 hours of
421 spreading. In contrast, Tiwary *et al.* (2015) found that NH_3 losses from food based digestates
422 continued for about 2 days following spreading (although NH_3 emission patterns in tropical climates
423 are likely to be different from those in temperate conditions), but that incorporation immediately
424 following application reduced NH_3 losses by 85%. These findings highlight the importance of rapid
425 soil incorporation as an alternative method for preventing N losses via this pathway on arable
426 (cultivated) soils. Rapid soil incorporation has previously been shown to reduce NH_3 losses following
427 slurry applications (e.g. Sommer & Hutchins, 2001; Webb *et al.*, 2005), although UK survey results
428 have shown that a significant proportion of cattle and pig slurry (*c.*65%) applied to tillage fields is
429 either not incorporated, or only incorporated more than 24 hours after spreading (Benford, 2016).
430 This study has demonstrated that rapid incorporation could have the same mitigation potential
431 when used with food-based digestate. However, the technique has also been shown to increase N_2O
432 emissions following slurry application as (Thorman *et al.*, 2011) suggesting that the potential for so
433 called 'pollution swapping' could also exist when it is used with digestates.

434 In addition to representing the loss of a valuable resource, NH_3 emissions from digestate applications
435 present a challenge to the UK meeting EU directives on NH_3 emissions. Under the EU National
436 Emissions Ceiling Directive, the UK has a proposed target to reduce ammonia emissions by 8%
437 (relative to a 2005 baseline) between 2020 and 2029, and by 21% from 2030. The UK Ammonia
438 Emissions Inventory (UKAEI) includes emissions from food and crop-based digestates (but in the non-
439 agricultural Inventory), although better data are needed to improve the current estimates. Based on
440 the emission factors measured in this study (i.e. 38% for bandspread food-based digestate; Table 3)
441 and the estimated 1.4 million m^3 of food-based digestate currently applied to agricultural land
442 (WRAP, 2014) with an average total N content of 5 kg/m^3, this implies that food-based digestate will
443 emit an additional 3.3 kt of NH_3, equivalent to 1% of the UK emission target (297 kt for 2010). If this
444 is scaled up to the predicted 2020 production volumes (4.25 million m^3), then food-based digestate
445 would emit an additional 9.8 kt of NH_3 or 3.3% of the UK 2010 target. However, these estimates do
446 not account for reductions in ammonia emissions from manufactured (mineral) N fertiliser use.

447 4.2 Nitrous oxide emissions

448   Nitrous oxide emissions from agricultural soil are predominately produced via the microbially

449   mediated processes of nitrification and denitrification (Firestone & Davidson, 1989). The factors

450   which control the magnitude of N_2O emission include soil mineral nitrogen (SMN) content, soil

451   temperature, soil moisture content (Dobbie & Smith, 2001; 2003) and available carbon (Weier *et al.*,

452   1993). Davidson (1991) showed that there is an optimum production of N_2O, which occurs at a

453   water-filled pore space (WFPS) roughly equal to a transition point below which N_2O is predominantly

454   emitted from the aerobic process of nitrification and above which N_2O is predominantly emitted

455   from the anaerobic process of denitrification. Davidson (1991) suggested that this transition occurs

456   at a WFPS of 60%; other studies, however, have shown that the position of the maximum emission

457   can vary with soil type and conditions. Notably, UK studies have indicated that the highest N_2O

458   emissions frequently occur as a result of the anaerobic process of denitrification i.e. at a WFPS >60%

459   (e.g. Dobbie & Smith, 2001; 2003).

460   At all the 3 experimental sites, most N_2O emissions occurred in the few weeks following the organic

461   materials being spread to land, and had generally returned to background levels within *c.*2 months

462   (Figures 2a, 3a, 4a and 5a). Tiwary *et al.* (2015) also reported that N_2O emission rates were "feeble"

463   past 24 hours following spreading of food-based digestate at a site in India, and emissions from soil

464   incorporated digestate were negligible; however they only measured emissions for 30 days.

465   At NW and WE, the highest N_2O emissions corresponded to a peak in soil NH_4-N concentrations (see

466   Supplementary Information, Figure S2) suggesting that N_2O was being produced as a result of the

467   nitrification of the NH_4-N in the applied organic materials to NO_3^--N by soil micro-organisms. In this

468   study, there was no clear relationship observed between N_2O emission rates and changes in the

469   WFPS (Figures 2a, 3a, 4a and 5a). However, at NW there was a peak in emissions corresponding to a

470   *c.*10% drop in WFPS which may have been caused by a change from anaerobic to aerobic conditions

471   which moved the end product of denitrification from N_2 to N_2O.

472   Importantly, N_2O emission factors (EFs) from all the organic material treatments were below the

473   IPCC Tier 1 default value of 1% of total N applied, and in the case of compost was not significantly

474   different from background values, although the variability associated with emissions from some

475   treatments suggests that the IPCC value Tier 1 default value could be exceeded on some occasions

476   (see Figures 3b and 5b). The EF for food-based digestate (mean 0.45% ± 15%) was much lower than

477   those obtained by Tiwary *et al.* (2015) of 4 – 10%, although the latter values were obtained under

478   tropical climatic conditions which are very different to the UK and may have promoted N_2O losses.

479   Data derived from the present study could be used in conjunction with other recently published

480   research results on N_2O emissions from fertilisers and manures (Bell *et al.,* 2015a,b; Bell *et al.,* 2016)

481   to reduce some of the uncertainty in the UK national agricultural N_2O inventory through the

482   generation of robust and experimentally verified Tier 2 EFs in compliance with the requirements of

483   the IPCC (IPCC, 1996; 2006).

484

485   4.3 Nitrate leaching

486   Nitrate leaching losses from autumn food-based digestate applications were *c.*15% of the total N

487   applied compared with the IPCC default value of 30% for all agricultural soil N sources (i.e. synthetic

488   fertilisers, manures, compost, sewage sludge, crop residues and other organic N sources), which is

489   used when calculating indirect N_2O losses from N applications to land. Research in the 1990s using

490   livestock manures showed that NO_3^- leaching losses can be greatly reduced by applying the materials

491   in spring compared with autumn applications (e.g. Chambers *et al*., 2000), and this has led to the

492   introduction of no-spreading periods for high available N materials being an integral part of NVZ

493   legislation in England, Scotland and Wales (SI, 2015; SSI, 2013; WSI, 2013). Despite this, recent UK

494   survey data have shown that significant percentages of biosolids (64%) and other non-farm organic

495   materials (32%) are still applied to winter sown crops in August, September and October (Benford,

496   2016) and hence may be prone to nitrate leaching losses. This is mainly for practical reasons due to

497   storage capacity pressures, and so that farmers can spread materials to land while soils are still

498   trafficable and before a crop is sown. The results from our study strongly suggest that, as for

499   livestock slurry and other high readily available N organic materials, farmers should be advised to

500   apply food-based digestate in the spring where practically possible, or in autumn to an actively

501   growing crop such as grass or oilseed rape which will take up available N from the soil so it will not

502   be lost via overwinter NO_3^- leaching.

503   5. Conclusion

504   **T**he results of this study have shown that N losses via NH_3 volatilisation following land spreading of

505   food-based digestate were high at both the arable and grassland sites (30-50% of total N applied).

506   This was due to the high readily available N content and high pH of the food-based digestate (mean

507   8.5), and the soil conditions at the time of spreading which affected the rate at which the materials

508   infiltrated into the soil matrix. Precision application (i.e. bandspreading) can reduce NH_3 emissions,

509   but the effectiveness of these techniques is dependent on the prevailing soil conditions. Because the

510   majority of the NH_3 losses occurred within 6 hours of spreading, it is important that where possible

511   farmers are encouraged to rapidly incorporate food-based digestate into the soil as a method for

512   conserving N so it can be utilised by the crop (assuming application to spring crops). In contrast, N_2O

losses from food-based digestate were low, with measured emission factors all less than the 1% IPCC default value (mean 0.45% ± 0.15%). There was no significant difference between N_2O losses from food-based digestate applied by surface broadcast or bandspread techniques; however, we would recommend that further research is undertaken on different soil types and under different climatic conditions to confirm this finding, as these are important factors controlling denitrification/nitrification processes. Autumn applications could potentially lead to a total of 75% of the applied N being lost by ammonia volatilisation (*c.*60%) and leaching (*c.*15%), constituting the loss of a valuable resource with consequent economic and environmental implications. This strongly suggests that farmers should be advised to apply these materials in the spring where practically possible. Emissions (NH_3, N_2O, NO_3^-) from green compost were all low, indicating that in these terms compost can be considered as a low risk material in terms of N losses, which can be used to build up soil long-term (organic) N reserves and to improve soil condition.

The information produced from this study has been used to develop best practice guidelines for digestate and compost use in the UK that seek to maximise crop nutrient utilisation and to minimise emissions to the environment (WRAP, 2016) Furthermore, the N_2O and NH_3 EFs derived in this study for both food-based digestate and composts could be used to improve the estimates of emissions from these sources in the UK GHG and Ammonia Emission Inventories, although emissions under other agroclimatic conditions are likely to be different.

6. Acknowledgements

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730 **Table 1. Baseline soil characteristics, rainfall and cropping at the experimental sites.**

Site code	Site name and location	Soil texture	Clay (%)	Total N (%)	Organic C (%)	pH	Annual rainfall (mm)	Cropping
NW	North Wyke (Devon, England)	Clay loam	38	0.4	6.9	5.5	1,031	Grassland
PW	Pwllpeiran (Ceredigionshire, Wales)	Clay loam	28	0.5	4.7	5.1	975	Grassland
WE	Wensum (Norfolk, England)	Sandy loam	11	0.2	1.3	6.7	594	Winter wheat

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733   **Table 2. Organic material analysis, application rates and N loadings at each site**

Determinand	Food-based digestate	Compost*	Slurry**	FYM**
Site: NW				
Dry matter (%)	5.1	60	6.1	20
Total N (kg/t fw)	8.0	14	2.6	5.8
NH_4-N (kg/t fw)	5.8	0.7	1.4	<0.1
NO_3^-N (kg/t fw)	<0.1	<0.1	<0.1	<0.1
NH_4-N (% total N)	73	5	56	<1
pH	8.1	8.5	8.2	9.0
Application rate (t/ha)	20	20	30	25
N loading (kg/ha)	160	271	77	144
Site: PW				
Dry matter (%)	6.1	51	4.9	24
Total N (kg/t fw)	5.4	7.0	2.2	4.9
NH_4-N (kg/t fw)	3.9	<0.1	1.2	<0.1
NO_3^--N (kg/t fw)	<0.1	<0.1	<0.1	0.4
NH_4-N (% total N)	72	1	53	8
pH	8.4	7.9	7.0	7.3
Application rate (t/ha)	20	20	30	25
N loading (kg/ha)	107	140	67	122
Site: WE autumn				
Dry matter (%)	5.4	54	2.3	24
Total N (kg/t fw)	7.8	11	3.0	8.1
NH_4-N (kg/t fw)	6.3	1.5	2.2	0.8
NO_3^--N (kg/t fw)	<0.1	<0.1	<0.1	0.1
NH_4-N (% total N)	82	14	75	9
pH	8.8	7.0	7.5	8.4
Application rate (t/ha)	32	20	41	30
N loading (kg/ha)	245	216	122	244
Site: WE spring				
Dry matter (%)	4.4	48	2.7	23
Total N (kg/t fw)	6.9	9.0	2.6	9.2
NH_4-N (kg/t fw)	6.2	0.2	2.2	0.2
NO_3^--N (kg/t fw)	<0.1	<0.1	<0.1	0.6
NH_4-N (% total N)	89	2	86	3
pH	8.7	8.3	8.0	7.4
Application rate (t/ha)	30	20	38	30
N loading (kg/ha)	207	181	98	277

734   fw = fresh weight
735   *Green compost at NW and PW; green/food compost at WE
736   **Cattle slurry and FYM at NW and PW; pig slurry and FYM at WE

737 **Table 3. Cross-site NH$_3$ and N$_2$O emission factors.** *Means labelled with different superscript letters*

738 *are significantly (P<0.05) different from each other.*

Treatment	Mean NH$_3$ EF	Mean N$_2$O EF
	(% total N applied)	
Food based digestate – surface broadcast	42[d]	0.47[b]
Food based digestate – bandspread	38[d]	0.43[b]
Livestock slurry – surface broadcast	31[c]	0.35[b]
Livestock slurry – bandspread	24[b]	0.55[b]
Livestock FYM	4.5[a]	0.28[b]
Compost	3.3[a]	<0.01[a]
SED	*3.3*	*0.12*
LSD	*6.7*	*0.24*

739 SED: Standard error of difference of means

740 LSD: Least significant difference of means (5% level)

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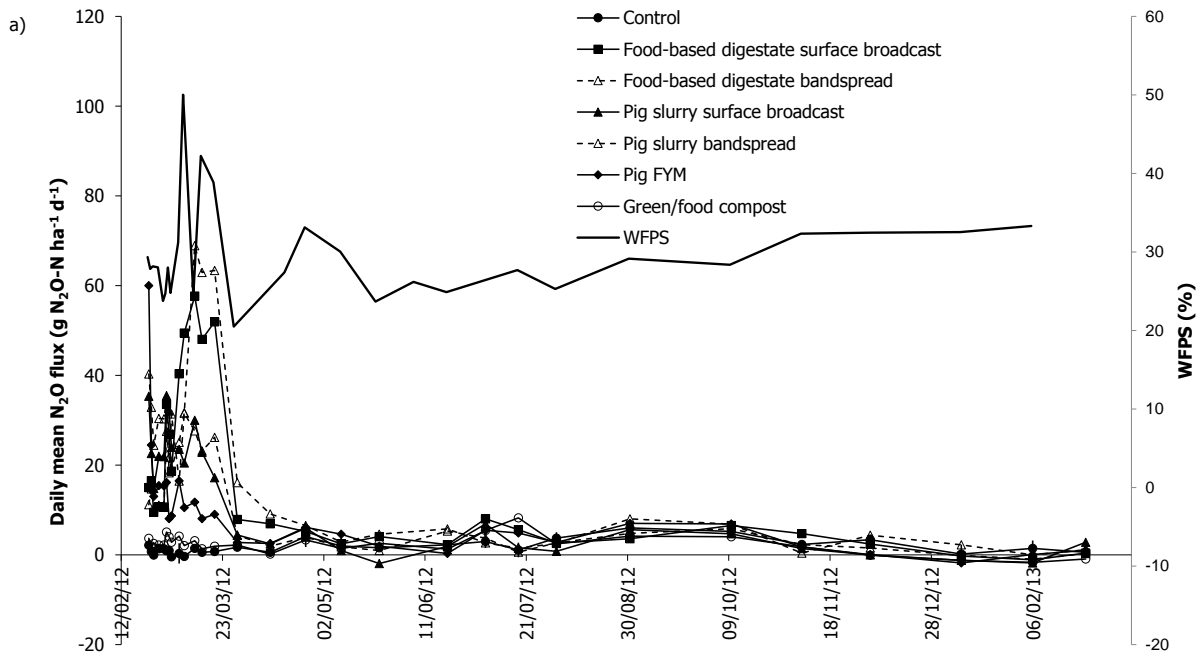
Figure 1. Cross-site ammonia emissions curves.

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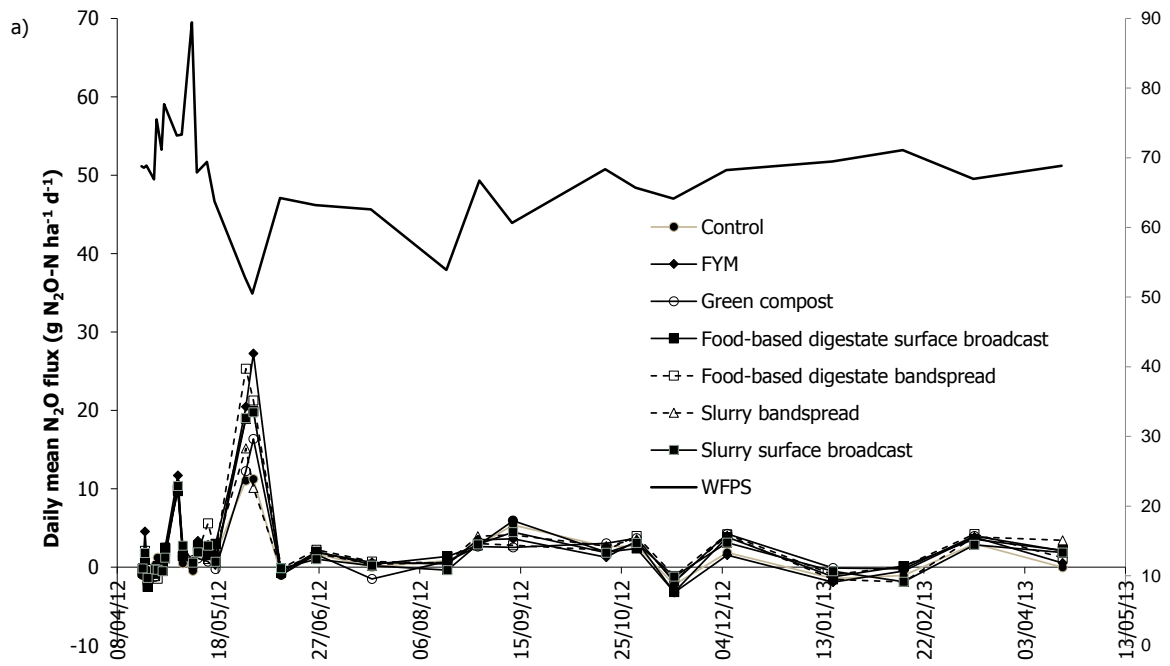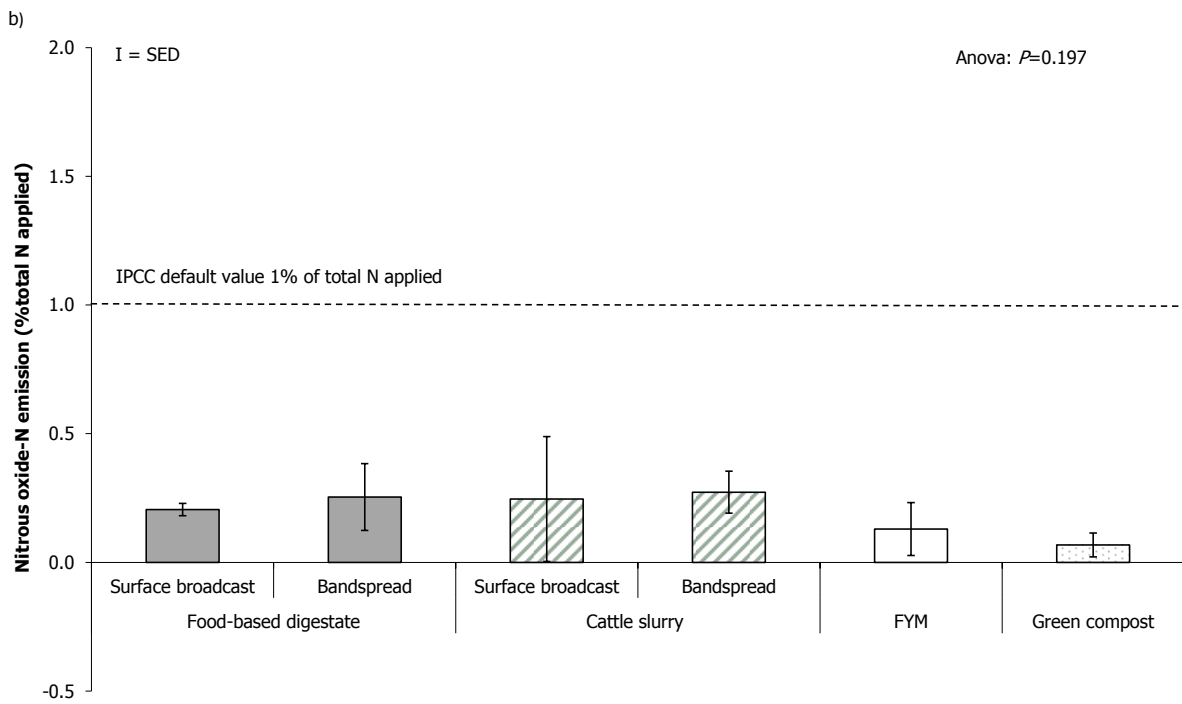
Figure 2. NW spring: a) daily mean N₂O fluxes and b) N₂O emission factors. *Error bars show the standard error; SED = standard error of difference of means. No significant treatment differences.*

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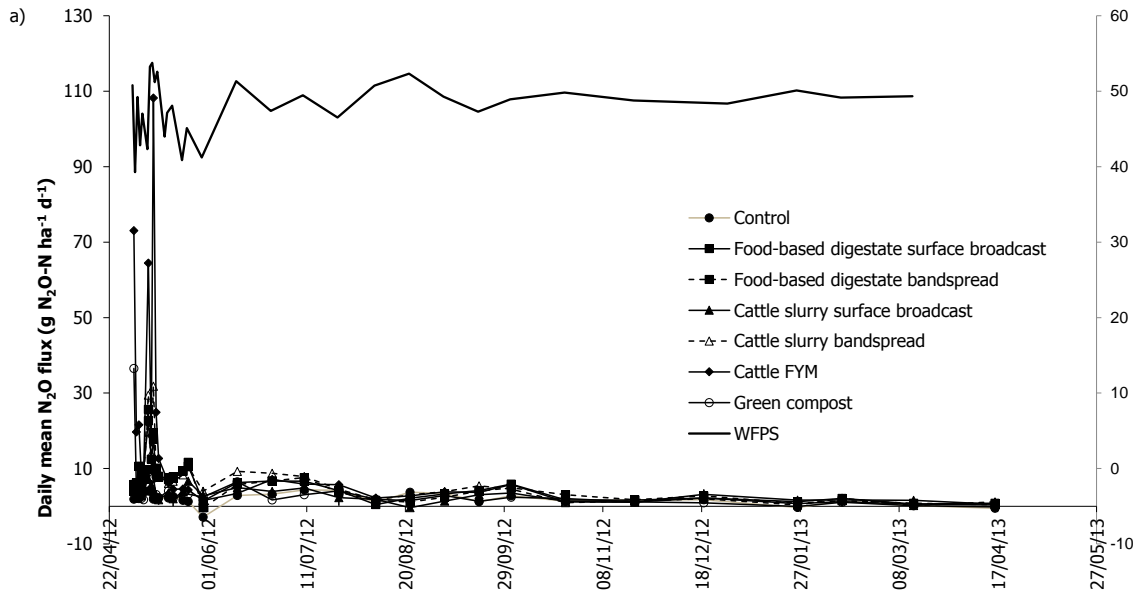
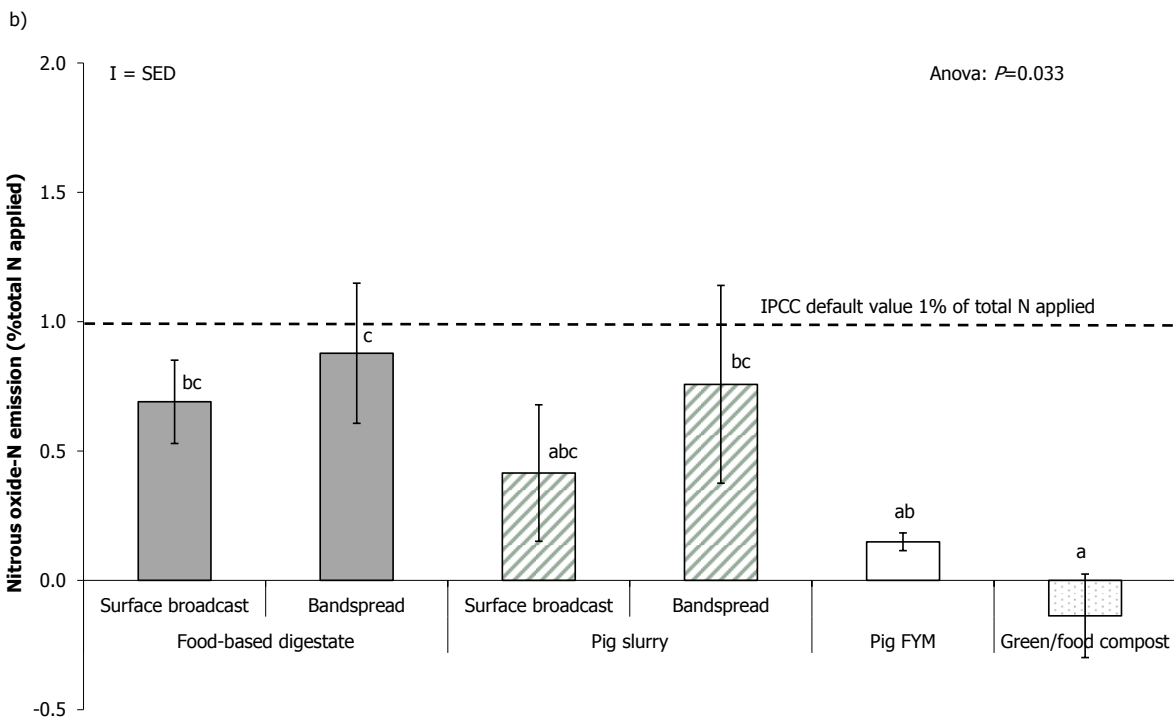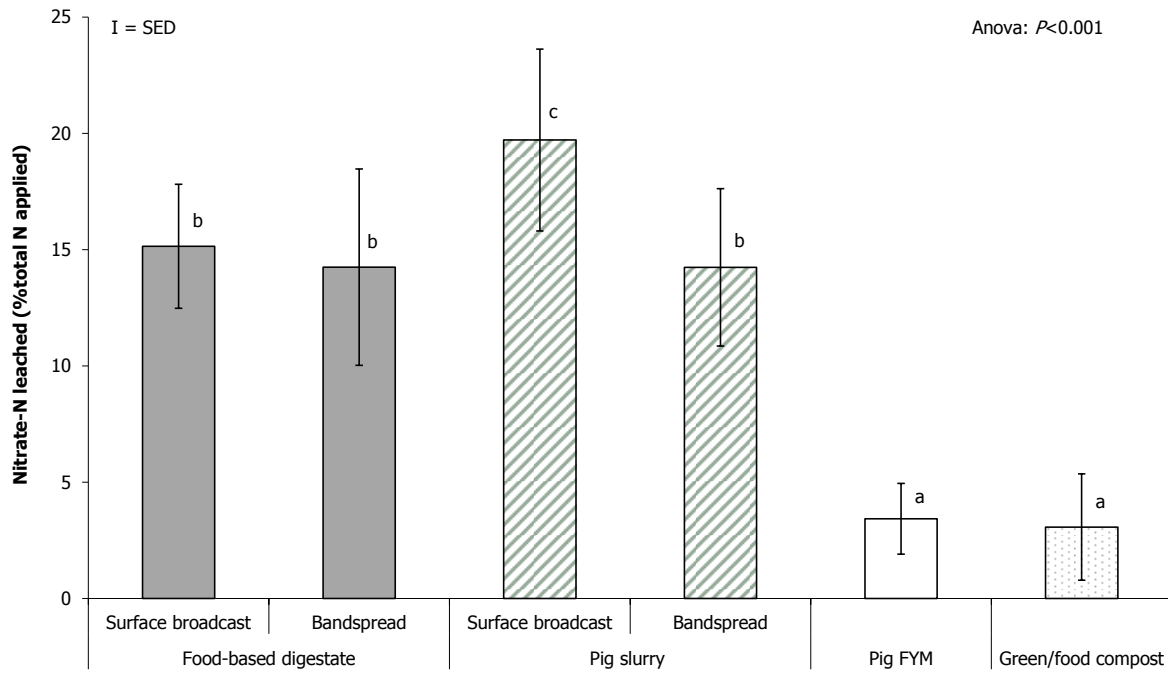764
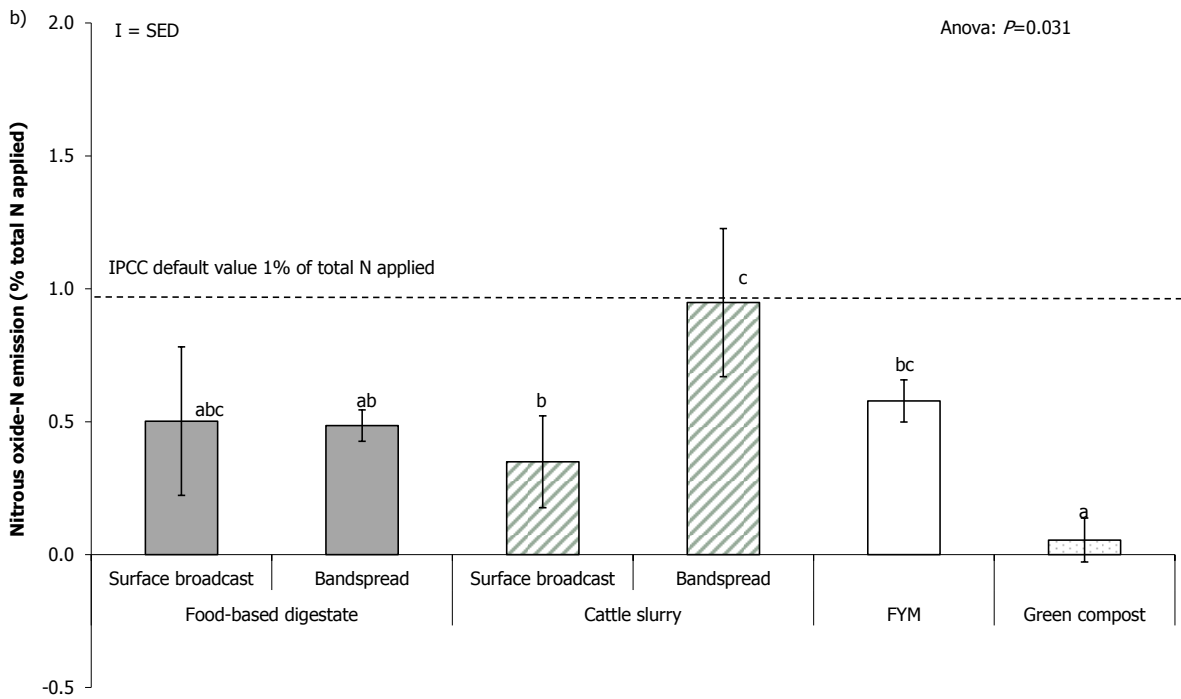
Figure 3. PW spring: a) daily mean nitrous oxide fluxes and b) emission factors. *Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly (P<0.05) different from each other.*

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a)

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b)
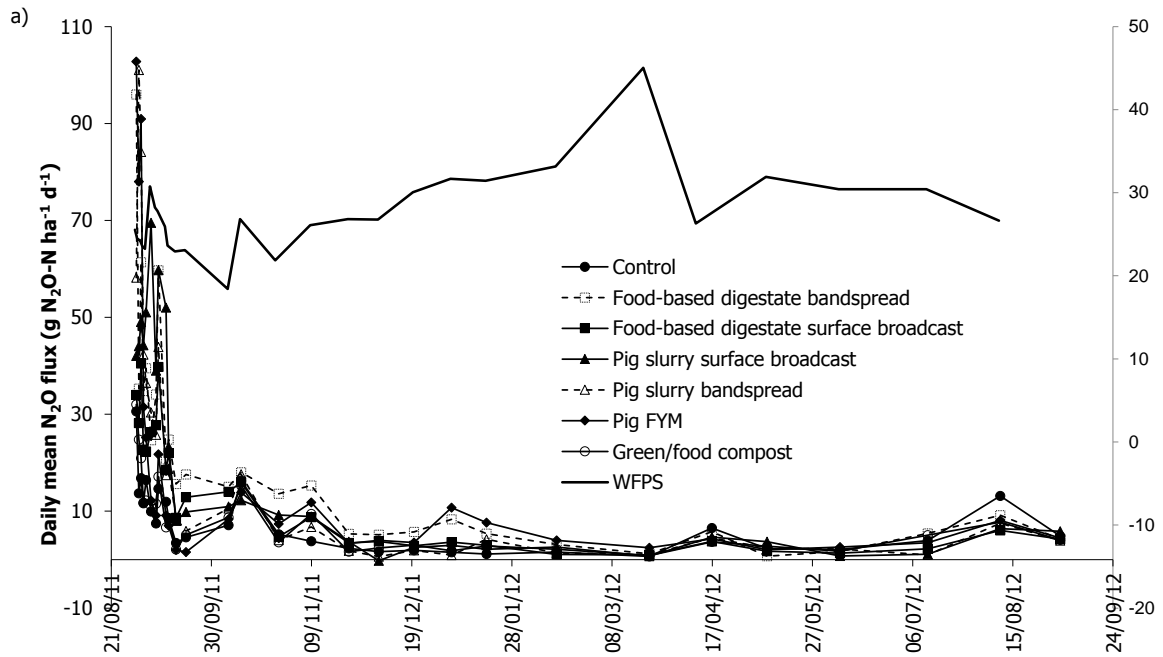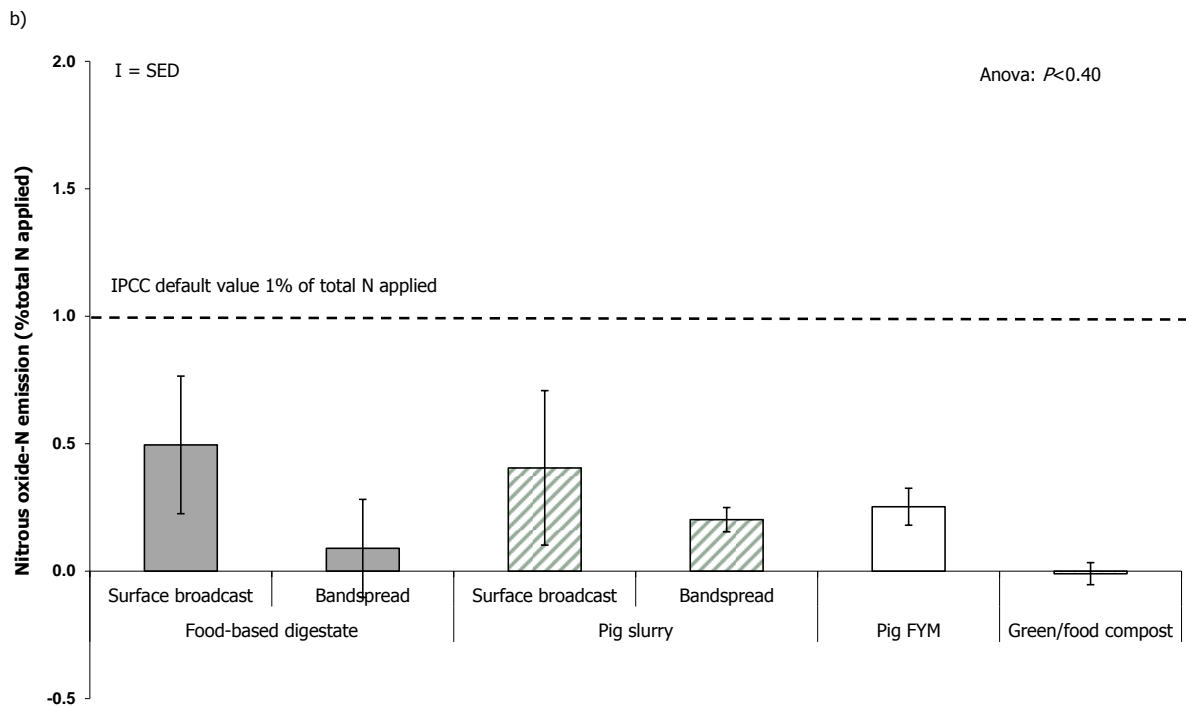
Figure 4. WE autumn: a) daily mean N$_2$O fluxes and b) N$_2$O emission factors. *Error bars show the standard error; SED = standard error of difference of means. No significant treatment differences.*

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Figure 5. WE spring: a) daily mean N₂O fluxes and b) N₂O emission factors. *Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly (P<0.05) different from each other.*
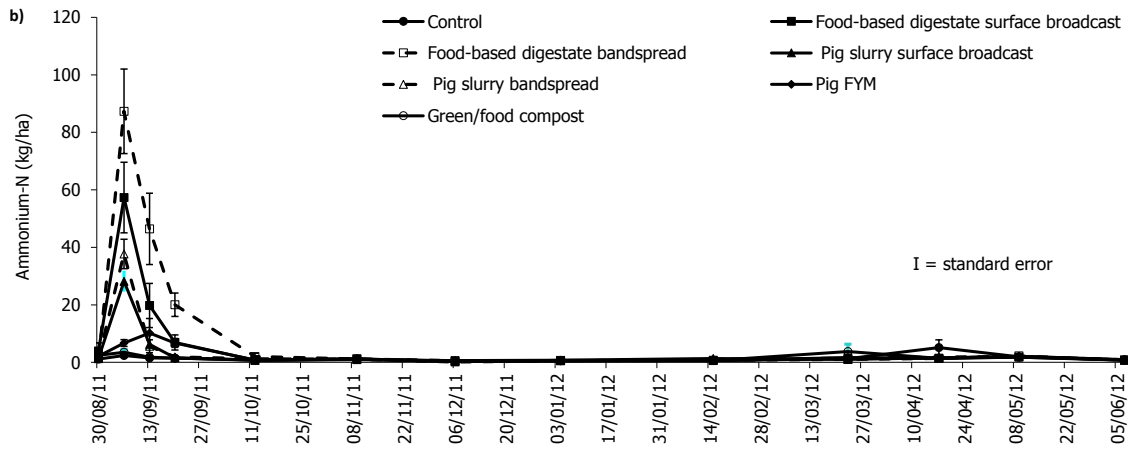
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Figure 6. Leaching losses (% of total N applied) following the autumn 2011 organic material applications. *Error bars show the standard error; SED = standard error of difference of means. Columns labelled with different letters are significantly (P<0.05) different from each other.*

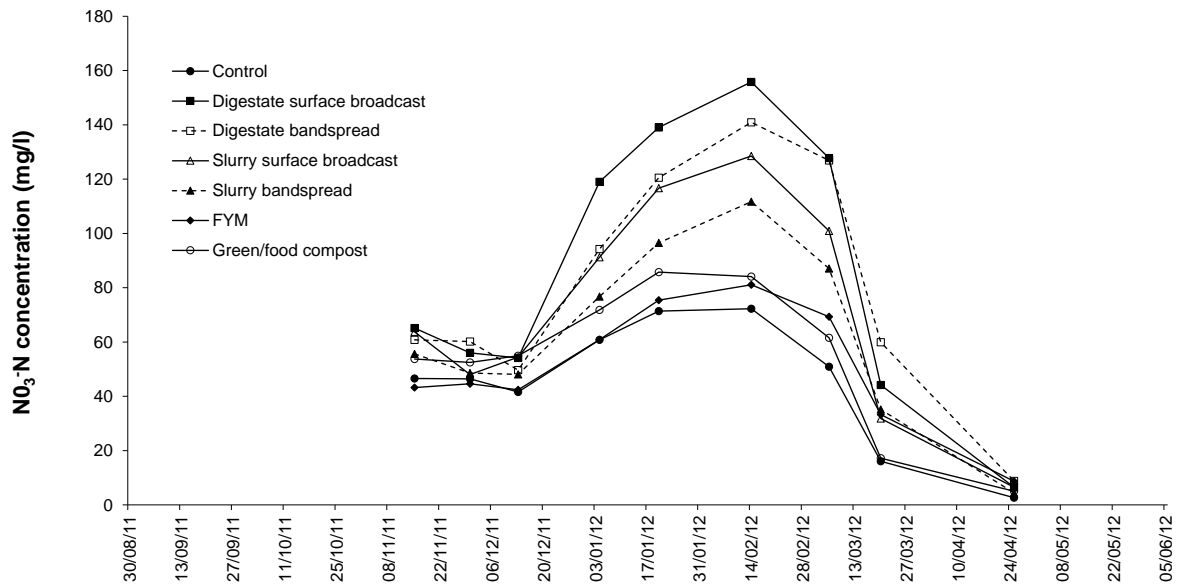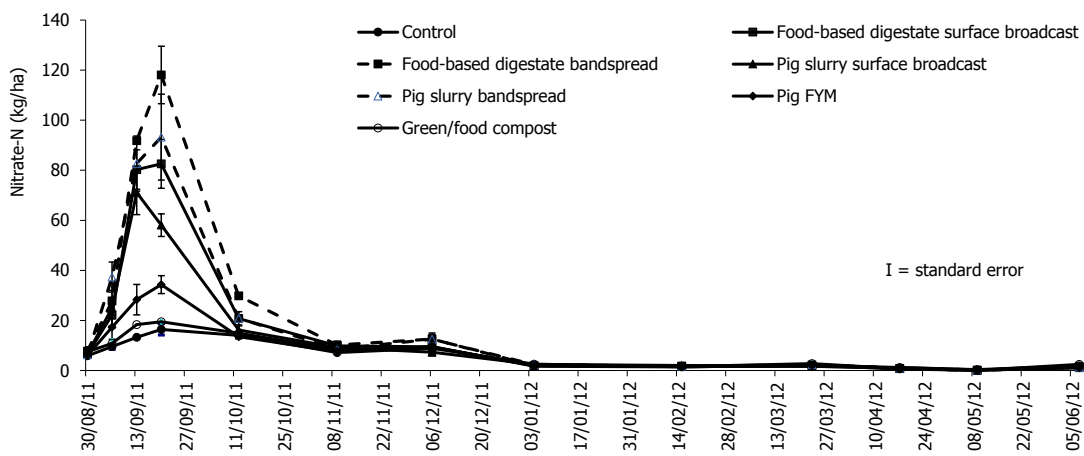775   **Supplementary Information**

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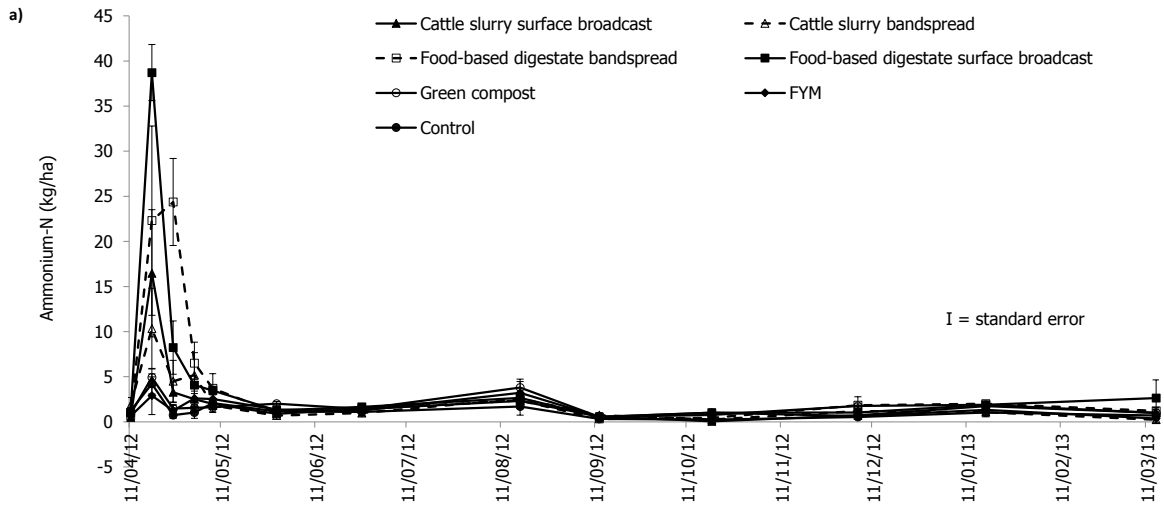
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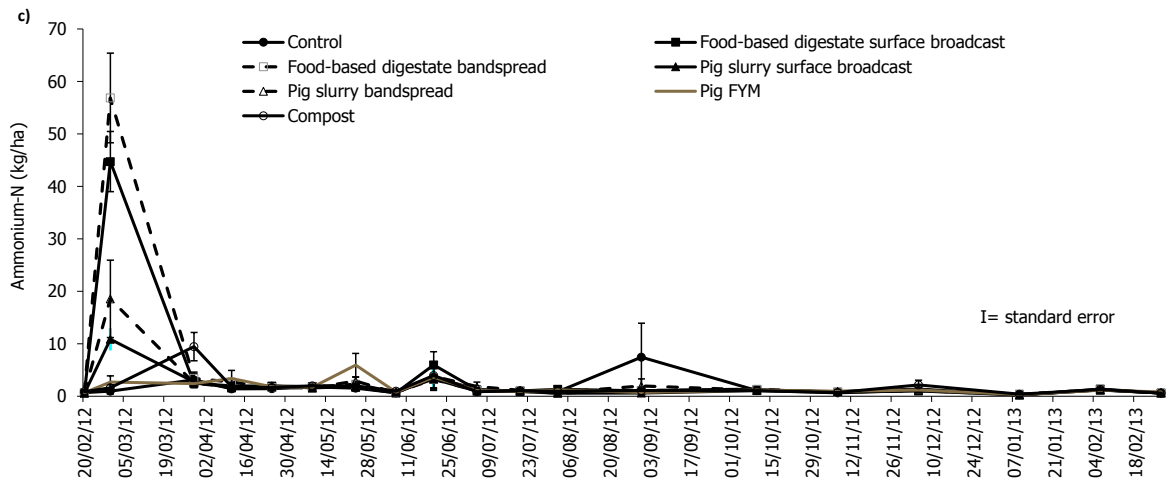
780   **Figure S1. Soil NH$_4$-N concentrations at a) NW, b) WE autumn and c) WE spring**

781   a)

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783   b)

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Figure S2. WE autumn a) NO$_3^-$ concentrations in leachate following the autumn 2011 organic material applications and b) soil NO$_3^-$ concentrations

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