

1 **Different approaches to assess the environmental performance of a cow manure**
2 **biogas plant**

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17

18 **Abstract**

19 In intensive livestock production areas, farmers must apply manure management
20 systems to comply with governmental regulations. Biogas plants, as a source of
21 renewable energy, have the potential to reduce environmental impacts comparing with
22 other manure management practices. Nevertheless, manure processing at biogas plants
23 also incurs in non-desired gas emissions that should be considered. At present, available
24 emission calculation methods cover partially emissions produced at a biogas plant, with

25 the subsequent difficulty in the preparation of life cycle inventories. The objective of
26 this study is to characterise gaseous emissions: ammonia ($\text{NH}_3\text{-N}$), methane (CH_4),
27 nitrous oxide ($\text{N}_2\text{O}_{\text{indirect}}$, and $\text{N}_2\text{O}_{\text{direct}}$) and hydrogen sulphide (H_2S) from the anaerobic
28 co-digestion of cow manure by using different approaches for preparing gaseous
29 emission inventories, and to compare the different methodologies used. The chosen
30 scenario for the study is a biogas plant located next to a dairy farm in the North of
31 Catalonia, Spain. Emissions were calculated by two methods: field measurements and
32 estimation, following international guidelines. International Panel on Climate Change
33 (IPCC) guidelines were adapted to estimate emissions for the specific situation
34 according to Tier 1, Tier 2 and Tier 3 approaches. Total air emissions at the biogas plant
35 were calculated from the emissions produced at the three main manure storage facilities
36 on the plant: influent storage, liquid fraction storage, and the solid fraction storage of
37 the digestate. Results showed that most of the emissions were produced in the liquid
38 fraction storage. Comparing measured emissions with estimated emissions, NH_3 , CH_4 ,
39 $\text{N}_2\text{O}_{\text{indirect}}$ and H_2S total emission results were in the same order of magnitude for both
40 methodologies, while, $\text{N}_2\text{O}_{\text{direct}}$ total measured emissions were one order of magnitude
41 higher than the estimates. A Monte Carlo analysis was carried out to examine the
42 uncertainties of emissions determined from experimental data, providing probability
43 distribution functions. Four emission inventories were developed with the different
44 methodologies used. Estimation methods proved to be a useful tool to determine
45 emissions when field sampling is not possible. Nevertheless, it was not possible to
46 establish which methodology is more reliable. Therefore, more measurements at
47 different biogas plants should be evaluated to validate the methodologies more
48 precisely.

49

50 **Keywords:** GHG emission; IPCC estimation; Ammonia emission; Livestock manure;
51 Biogas plant; Emission inventory.

52

53 **1. Introduction**

54 Manure management is one of the greatest challenges for farmers. In intensive livestock
55 production areas, an appropriate management of manure is necessary, not only to
56 comply with legislation –for instance, when it comes to groundwater pollution by
57 nitrates from agricultural sources (EEC, 1991; Generalitat de Catalunya, 2009),
58 ammonia emission ceilings (EEC, 2001) or the use of renewable energy targets for
59 reducing greenhouse gases (EEC, 2009)–, but also as a matter of social concern. On the
60 other hand, in the framework of a circular economy, manure should be considered a
61 valuable resource. From this point of view, manure processing technologies could be
62 helpful to close nutrient cycles while reducing emissions to the environment. The
63 implementation of technologies designed to protect the environment is variable across
64 Europe but those methods perceived as good farming practice, or those which benefit
65 the farmer in other ways, are the most widespread (Loyon et al., 2016). The selection of
66 a particular manure processing technology depends, among other things, on
67 technological knowledge, implementation feasibility, local constraints, economic
68 viability, and farm management strategy (Flotats et al., 2011).

69 In this context, biogas plants are considered as an efficient solution to improve manure
70 management and to mitigate pollution problems while producing renewable energy
71 (Massaro et al., 2015). Nevertheless, manure processing at biogas plants also incurs in
72 non-desired gas emissions. The sustainability of anaerobic digestion plants has been

73 analysed from the perspective of life cycle assessment (Cherubini et al., 2015; Lijó et
74 al., 2014; Poeschl et al., 2012a, b). A LCA requires a detailed inventory for the analysed
75 system, compiling all inputs and outputs in the product life cycle. The assessment of
76 emissions is a key point in the preparation of an inventory and a variety of methods are
77 available to calculate emissions. It is necessary to pay attention to the selection of the
78 method of emission calculation in order to assess the amount of emissions from manure
79 management systems (Braschel and Posch, 2013). On the other hand, emission factors
80 can be determined at laboratory and pilot scale experiments (Perazzolo et al., 2015),
81 and both emission measurements and estimations can be used to gain knowledge about
82 emissions produced at field scale (Owen and Silver, 2015; Vac et al., 2013).

83 Field emission measurement presents both advantages and drawbacks. Emission
84 sampling provides the amount of gas released for specific climatic conditions and local
85 geographic characteristics (Sommer et al., 2000). However, emission sampling is an
86 expensive and time-consuming technique. Due to physical and structural constraints, it
87 might be difficult to obtain a representative characterization of all emission
88 points. Therefore, when sampling of emissions is not affordable, an estimation, using
89 normalized methods, can be useful (Storm et al., 2012).

90 International organisations provide guidebooks to compile an atmospheric emission
91 inventory, (EEA, 2013; IPCC, 2006). Different estimation proxies can be reached
92 depending on data availability and the complexity of the methodology applied. The
93 IPCC guidebook (IPCC, 2006) defines Tier 1, Tier 2, and Tier 3; being Tier 1 the
94 simplest method and Tier 3 the most complex one. These guidebooks provide
95 algorithms and emission factors to estimate, among others, CH₄, NH₃ and N₂O
96 emissions from some of the most usual manure management systems (e.g. storage

97 ponds, composting, etc.), but they are not detailed for other increasingly common
98 practices as by-product storage from a processing plant. After manure treatments,
99 processed manure may change its physical and chemical characteristics. For instance,
100 during anaerobic digestion, easily degradable C is consumed and thus, less organic
101 matter is available to produce “uncontrolled” CH₄, which leads to lower CH₄ emission
102 capacities from the digestate than from untreated slurry (Rodhe et al., 2015). Therefore,
103 it is necessary to develop and adapt methodologies for such specific situations in order
104 to estimate the emissions occurring in these processing plants.

105 The main objective of the present study was to compare different approaches to
106 characterise gas emissions from a biogas plant processing cow manure and industrial
107 food waste. For this purpose, different methods were used and compared: field sample
108 measurement and emission estimation, following the IPPC adapted guidelines.

109 Additionally, a simplified procedure to estimate air emissions from the storage of
110 anaerobic digestion of manure by-products is provided to prepare a biogas plant
111 emission inventory.

112

113 **2. Materials and methods**

114 The emissions included in this study were those produced during the storage and
115 processing of manure. The characterisation of emissions was done by field emission
116 measurement and by the calculation of estimates. The emissions selected for the
117 characterisation of the biogas plant were ammonia (NH₃), methane (CH₄) and nitrous
118 oxide (N₂O) gases, due to their relevance in manure management systems and their
119 contribution to environmental impact. Hydrogen sulphide (H₂S) was also considered,
120 due to its contribution to acidification and potential for unpleasant odour generation.

121

122 **2.1 Processing plant description**

123 The scenario for the case in study was a dairy farm located in the North of Catalonia,
124 Spain, with an average of 920 cattle heads. The farm included two distinct areas: the
125 farm itself, with all the dairy cattle facilities, and a biogas plant. The slurry was stored
126 in the influent storage covered pond (129 m³). The co-substrate was stored in a covered
127 pond (59 m³) next to the influent storage pond. The co-substrates used in the plant were
128 mainly organic waste from food industry and sewage sludge from wastewater treatment
129 plants. The addition of co-substrate, which was below 20 %, aimed to increase biogas
130 production. The influent was estimated to remain in the influent storage pond (64 m²)
131 for around two days. Then, the slurry was pumped to the anaerobic digesters (AD), first
132 to two primary AD (AD1' and AD1'', 2,078 m³ each) and then from the primary to the
133 secondary AD (AD2, 1,450 m³). Digesters consisted of continuous stirred-tank reactors
134 (CSTR). The anaerobic digestion of slurry and co-substrate was conducted at
135 mesophilic conditions (37 °C) with a hydraulic retention time (HRT) of about 50 days.
136 Biogas was conducted to the CHP engine (500 kW) to produce electrical and thermal
137 energy. This electricity was sold to the grid and the thermal energy was used to heat the
138 reactors. The digestate was mechanically separated with a screw press separator,
139 producing a liquid and a solid fraction. The liquid fraction was stored at an un-covered
140 storage pond (2015 m²; 10,000 m³) during six months, and the solid fraction was
141 composted in trenches (4 trenches of 3.5 m · 20 m · 1.5 m each) for four months. A gas
142 purification system to eliminate H₂S from the biogas is installed before the CHP engine.

143

144 **2.2 Sampling procedure**

145 Emission sampling was carried out with a total of eight sampling campaigns. The
146 sampling days were programmed so that the samples were collected coinciding with the
147 day after the liquid fraction storage pond was filled and the day before the pond was
148 emptied. Therefore, samples were collected every two months approximately. This
149 sampling schedule was carried out to give insight into the specific management of the
150 treatment plant evaluated.

151

152 **2.2.1 Emission sampling and calculations**

153 Field emission measurements were made at the influent storage tank (IS) and at the
154 digestate liquid and solid fraction final storages, namely the liquid fraction storage
155 compartment (LFS) and the solid fraction storage compartment (SFS) corresponding to
156 the compost piles.

157 All samplings were conducted in the morning, between 10:00 and 14:00. Climate
158 conditions in the area were collected from the nearest weather station in Banyoles
159 (Meteocat, 2016). Detailed information on temperature, humidity, rainfall and wind for
160 each sampling day is presented in Table S1 (supplementary material). Air samples were
161 collected with a floating wind tunnel set up following the Lindvall Hood Method (EPA,
162 2001). The Lindvall Hood and other similar dynamic chambers are used not only for
163 odour measurements but also for other emissions sampling (Misselbrook et al., 1998;
164 Rostami et al., 2015; van Belois and Anzion, 1992). It should be mentioned that the
165 Lindvall Hood method could overestimate emissions as the method modify the local
166 conditions that affect the release of gases. The hood surface area was 0.92 m^2 . In every
167 sampling campaign the tunnel was placed in the same point. When placed over the
168 liquid surface, the chamber sank few centimetres into the pond forming a seal between

169 the surface and the air within the wind tunnel, avoiding gas losses. In the case of the
170 compost piles, the chamber should penetrate also several centimetres into the compost
171 in order to seal it. A ventilator was connected to the inlet opening of the hood to drive a
172 constant air flow inside the hood. The airflow velocity in the wind tunnel was set
173 following EPA recommendations for the Lindvall Hood (EPA, 2001), which must be
174 between $0.2 - 0.3 \text{ m s}^{-1}$. Before emission measurements, airflow velocity was measured
175 with an anemometer and airflow velocity was adjusted accordingly. When calculating
176 emissions, the value of the background concentration in the atmosphere was subtracted.
177 For each of the analysed compounds the sampling procedure was different and is
178 detailed hereunder.

179 *Ammonia nitrogen ($\text{NH}_3\text{-N}$) emissions* were trapped bubbling a known volume of air
180 into two serial glass midget impingers containing 10 mL of sulphuric acid 0.1 N. Three
181 replicates per point were taken for five minute sampling time periods, each one of them
182 at a flow rate of 1 L min^{-1} , approximately.

183 *Hydrogen sulphide (H_2S) emissions* were sampled using a prefilter of $0.45 \mu\text{m}$ of pore
184 size (Zefluor Polytetrafluoroethylene), supported by a porous plastic support pad, 25
185 mm of diameter, followed by a 10 cm glass tube containing 400/200 mg of coconut
186 shell charcoal, for a 20 minute sampling time period, at a flow rate of 0.2 L min^{-1} . Two
187 replicates per point were collected in this case.

188 *Methane (CH_4) and direct dinitrogen oxide ($\text{N}_2\text{O}_{\text{direct}}$) emissions:* Sampling of
189 greenhouse gases was carried out using an air sampling pump, connected to a 3 litre air
190 sample bag (SamplePro FlexFilm) at a flow rate of 1 L min^{-1} . Just one replicate per
191 point was taken but samples were analysed by duplicate.

192 To calculate the specific emission rates, firstly it is necessary to know the “hood
193 constant”, which can be calculated as:

$$194 \quad L = F_{path} / A \quad (1)$$

195 where L is the hood constant, F_{path} is the flow path cross section area (m^2) and A is the
196 covered area (m^2). In this case the hood constant was 0.0085, F_{path} was 0.0079 m^2 and A
197 was 0.9216 m^2 .

198 Once the hood constant was known, the specific emission rates (E_{sp} , $\text{mg m}^{-2} \text{ s}^{-1}$) could
199 be calculated from the concentration measured at the exit of the hood (C_{hood} , mg m^{-3})
200 and the flow velocity (V , m s^{-1}), as follows with equation 2:

$$201 \quad E_{sp} = C_{hood} \cdot L \cdot V \quad (2)$$

202 *Indirect dinitrogen oxide ($N_2O_{indirect}$) emissions* result from volatile nitrogen losses that
203 occur primarily in the forms of ammonia and NO_x (IPCC, 2006). Indirect N_2O
204 emissions could not be measured and they were calculated with equation 5, see section
205 2.4.

206

207 **2.2.2 Manure sampling and mass balance**

208 In order to characterise the performance of the biogas plant, liquid and solid manure
209 samples were collected on a monthly basis and characterized, carrying out a mass and
210 component balance of the entire plant. Samples were taken from the influent storage
211 tank after 60 minutes of intense mixing, the effluent from AD2, the liquid and solid
212 fractions of digestate after solid-liquid separation, LFS, and SFS. In the three storage
213 ponds where emission measurements took place, manure samples were collected next to
214 where the Lindvall Hood was placed. Solid samples were collected from different points
215 of the compost pile at 20-30 cm depth. One replicate per point was randomly taken.

216 The yearly component mass balance for each of the solid and liquid streams (total
217 solids, volatile solids, etc.) were calculated taking into account: the influent mass flow
218 of the treatment plant (data provided by the biogas plant operators); the removal
219 efficiency in the anaerobic digestion (AD was considered to remove 10% of the mass
220 due to TS degradation and the subsequent CH₄ production); the separator efficiency
221 measured in the plant; and the average composition of each stream.

222

223 **2.3 Analytical methods**

224 To determine the amount of ammonium nitrogen (NH₃-N) emissions, samples were
225 analysed using a modified method based on Conditional Test EPA Method (CTM-027)
226 sampling (EPA draft, 1997), and colorimetric NIOSH Method 6015 analysis (NIOSH,
227 1994b). H₂S emissions samples were desorbed from solid sorbent tubes with eluent, and
228 the resulting sulphate analysed by ionic chromatography following the NIOSH 6013
229 Method instructions (NIOSH, 1994a). The amount of CH₄ was determined using a
230 THERMO TRACE 2000 (Thermo Fisher Scientific, USA) gas chromatograph equipped
231 with a flame ionization detector (FID). N₂O emissions were determined with an Agilent
232 7820A (Agilent, USA) GC system using an electron capture detector (ECD).

233 Standard methods were used to analyse total Kjeldahl nitrogen (TKN) (4500-Norg C),
234 ammonium nitrogen (NH₄⁺-N) (4500-NH3 B C), pH (4500- H+ B), total solids (TS)
235 (2540 B) and volatile solids (VS) (2540 E), (APHA, 2014), and the chemical oxygen
236 demand (COD) following the method described in Nogueroles-Arias et al. (2012).

237 Potassium (K) was analysed by ionic chromatography following Standard Method 4110

238 B. Total phosphorus (P) was analysed by colorimetric Standard Method 4500-P C.

239 *Anaerobic biodegradability test*: In order to determine the methane producing capacity
240 (B_0) of manure, anaerobic biodegradability tests of the digestate as well as the liquid
241 and solid fractions of the digestate were performed –after mechanical separation–
242 following the methodology described in Silvestre et al. (2011).

243

244 **2.4. Estimation of emissions – IPCC method**

245 The International Panel on Climate Change (IPCC) guidelines (IPCC, 2006) were used
246 as guidance to estimate CH_4 , N_2O and NH_3 -N emissions, even if equations were adapted
247 to the present case in study.

248 The equations and emission factors to calculate emissions were the following:

249 *Methane (CH_4) emissions* (EF_{CH_4} , $kg\ CH_4\ y^{-1}$) were estimated with equation 3:

$$250\ EF_{CH_4} = VS \cdot B_0 \cdot 0.67 \cdot MCF / 100 \quad (3)$$

251 where, EF_{CH_4} is the CH_4 emission factor ($kg\ CH_4\ y^{-1}$), VS ($kg\ VS\ y^{-1}$) stands for volatile
252 solids excreted or contained in the manure, B_0 ($m^3\ CH_4\ kg\ VS^{-1}$) is the maximum VS
253 methane production capacity, 0.67 is the conversion factor of $m^3\ CH_4$ to $kg\ CH_4$, and MCF
254 (%) is the methane conversion factor for each manure management system according to
255 the climate region.

256 According to IPCC definitions of manure management systems, slurry storage previous
257 to anaerobic digestion corresponds to the system *Liquid/Slurry*. However, the digestate
258 storage compartment could not be appropriately defined as *Liquid/Slurry* because after
259 anaerobic digestion the characteristics of manure change significantly and may produce
260 methane at a different rate. Therefore, specific B_0 values, as a result of the anaerobic
261 biodegradability tests, were established to estimate methane emissions from fresh slurry
262 and digestate storages.

263 *Nitrous oxide (N₂O) direct emissions* (N₂O_{direct}, kg N₂O y⁻¹) were calculated following
 264 equation 4.

$$265 \quad N_{2O_{direct}} = N_{ex(T)} \cdot EF \cdot 44 / 28 \quad (4)$$

266 where $N_{ex(T)}$ (kg N y⁻¹) is the total amount of N excreted or contained in the manure, EF (kg
 267 N₂O-N kg N⁻¹) is the emission factor for the type of manure management system, and $44/28$ is
 268 the N₂O-N emissions to N₂O emissions conversion factor.

269 *Nitrous oxide (N₂O) indirect emissions* (kg N₂O y⁻¹) were estimated using Equation 5.

$$270 \quad N_{2O_{indirect}} = N_{volatilization} \cdot EF \cdot 44 / 28 \quad (5)$$

271 where $N_{volatilization}$ (kg NH₃-N + NO_x-N volatilised) is the total amount of volatilised N,
 272 EF is the emission factor for indirect N₂O emissions (default value is 0.01 kg N₂O-N
 273 (kg NH₃-N + NO_x-N volatilised)⁻¹ according to IPCC, Table 1), and $44 / 28$ is the N₂O-
 274 N emissions to N₂O emissions conversion factor.

275 *Ammonia (NH₃) emissions* (kg NH₃-N + NO_x-N volatilised) were calculated with
 276 equation 6:

$$277 \quad N_{volatilization} = N_{ex} \cdot Frac / 100 \quad (6)$$

278 where N_{ex} (kg N y⁻¹) is the total amount of N excreted or contained in the manure and
 279 $Frac$ (%) is the fraction of nitrogen that volatilises.

280 Depending on the specific livestock information and the complexity of the analysis,
 281 three approaches can be followed to estimate emissions from manure management,
 282 namely Tier 1, Tier 2, and Tier 3.

283

284 **2.4.1 Tier 1 approach**

285 In this approach, the characterisation of manure was done using IPCC default data:

286 Nitrogen excreted per dairy cattle in Western Europe (100 kg N head⁻¹ year⁻¹) and

287 volatile solid content in manure (5.1 kg VS head⁻¹ day⁻¹) (IPCC, 2006). The model was

288 adapted to calculate CH₄ emissions considering a 100 % slurry storage management
289 system and, for that system, selecting MCFs according to the average annual
290 temperature.

291 To calculate emissions of by-product storage, emission factors from literature and
292 several other considerations were taken into account: 1- Different manure storage time
293 periods before and after anaerobic digestion; 2- Total nitrogen content in manure was
294 the same before and after anaerobic digestion (Rodhe et al., 2015). Nitrogen emissions
295 at the influent storage pond and at the final storage pond were related to the manure
296 residence time period in the pond; 3- Total nitrogen and volatile solids content in the
297 solid fraction after screw press separation were 6 % and 23 %, respectively (Møller et
298 al., 2002); 4- Ammonia losses in the liquid fraction of the digestate were considered to
299 be of 40 %, just like in the raw manure storage compartment, and losses in the solid
300 fraction to be of 30 % (IPCC, 2006); 5- Volatile solids reduction in anaerobic digestion
301 was estimated at 45 % (El-Mashad and Zhang, 2010); 6- The maximum methane
302 producing capacity for the digestate was 121 m³ CH₄ kg VS⁻¹ (Rodhe et al., 2015).

303

304 **2.4.2 Tier 2 approach**

305 In this case, the characterisation of excreted manure was done using regional data from
306 Catalonia. Nitrogen excreted per dairy cattle was estimated at 80.22 kg N head⁻¹ year⁻¹
307 (Generalitat de Catalunya, 2009) and the volatile solid content in manure was 4.45 kg
308 VS head⁻¹ day⁻¹. To calculate CH₄, NH₃ and N₂O emissions, the same considerations
309 used in Tier 1 were taken into account.

310

311 **2.4.3 Tier 3 approach**

312 The characterisation of manure was done using analytical data. The total excreted
 313 nitrogen resulted in 148,109 kg N y⁻¹ and total VS in 2,318,869 kg VS y⁻¹. Maximum
 314 methane producing capacities were obtained from the biodegradability test (B₀, in m³
 315 CH₄ kg VS⁻¹). The NH₃ and N₂O emission factors used were the same as those used in
 316 the Tier 1 and Tier 2 approaches.

317 A summary of all considered factors and hypothesis used to estimate emissions is
 318 presented in Table 1.

319

320 **Table 1.** Emission factors (EF) used to estimate NH₃-N, CH₄ and indirect N₂O
 321 emissions in biogas plant compartments: influent storage (IS), liquid fraction storage
 322 (LFS) and solid fraction storage (SFS). Data sources are presented for Tier 1, Tier 2
 323 and Tier 3 approaches.

Emission	Storage	Factor	Tier 1	Tier 2	Tier 3	
NH ₃ -N	IS	N, available (kg N y ⁻¹)	86,613 ^a	69,880 ^b	152,642 ^c	
		Storage time (yearly %)	1.3	1.3	1.3	
		EF (% NH ₃ -N+NO _x -N loss)	40 ^a	40 ^a	40 ^a	
	LFS	N, available (kg y ⁻¹)	81,407 ^{a+e}	65,681 ^{b+e}	73,779 ^c	
		N separator efficiency (%)	94.0 ^e	94.0 ^e	-	
		Storage time (yearly %)	98.7	98.7	-	
		EF (% NH ₃ -N+NO _x -N loss)	40 ^a	40 ^a	40 ^a	
		SFS	N, available (kg y ⁻¹)	5,205 ^{a+e}	4,200 ^{a+e}	3,557 ^c
			N separator efficiency (%)	6.0 ^e	6.0 ^e	-
EF (% NH ₃ -N+NO _x -N loss)	30 ^a		30 ^a	-		
CH ₄	IS	VS, available (kg VS y ⁻¹)	1,717,234 ^{a+d}	1,498,371 ^{b+d}	2,910,802 ^c	
		Storage time (yearly %)	1.3 ^d	1.3 ^d	-	
		MFC CH ₄ (%)	27; 29 ^a	27; 29 ^a	27; 29 ^a	
		B ₀ (m ³ CH ₄ ·kg VS ⁻¹)	0.24 ^a	0.24 ^a	0.273 ^f	
		LFS	VS, available (kg VS y ⁻¹)	727,437 ^{a+e+g}	634,725 ^{b+e+g}	519,989 ^c
	VS post AD (%)		55 ^g	55 ^g	-	
	VS separator efficiency (%)		77.0 ^e	77.0 ^e	-	
	MFC CH ₄ (%)		27; 29 ^a	27; 29 ^a	27; 29 ^a	
	B ₀ (m ³ CH ₄ ·kg VS ⁻¹)		0.121 ^h	0.121 ^h	0.086 ^f	

	SFS	VS, available (kg VS y ⁻¹)	217,041 ^{a+e+g}	189,379 ^{b+e+g}	103,832 ^c
		VS post AD (%)	55 ^g	55 ^g	-
		VS separator efficiency (%)	23.0 ^e	23.0 ^e	-
		MFC CH ₄ (%)	4.0 ^a	4.0 ^a	4.0 ^a
		B ₀ (m ³ CH ₄ ·kg VS ⁻¹)	0.121 ^h	0.121 ^h	0.100 ^f
N ₂ O _{indirect}	IS, LFS, SFS	N ₂ O EF (kg N ₂ O-N (kg NH ₃ -N+NO _x -Nvol) ⁻¹)	0.01 ^a	0.01 ^a	0.01 ^a
N ₂ O _{direct}	SFS	N, available at SFS (kg y ⁻¹)	5,205 ^{a+e}	4,200 ^{b+e}	3, 557 ^c
		N separator efficiency (%)	6.01 ^e	6.01 ^e	-
		Direct N ₂ O EF (kg N ₂ O-N kg N total ⁻¹)	0.005 ^a	0.005 ^a	0.005 ^a

324

325 ^aIPCC, 2006; ^bRegional data, Generalitat de Catalunya, 2009; ^cAnalytical data; ^dAssumption; ^eMoller et
 326 al., 2002; ^fLaboratory test; ^gEl Mashad et al., 2010; ^hRodhe et al., 2015.

327

328 2.5. Uncertainty analysis

329 An uncertainty analysis was carried out for manure composition parameters and
 330 emissions obtained from analytical data. Standard deviation was included in results for
 331 mass and component balance of manure samples, emission rates, emission field
 332 measurements and emission Tier 3 estimates, as these were the results obtained from
 333 experimental data.

334 Measured and Tier 3 estimated emissions were analysed with an additional statistical
 335 test in order to assess the uncertainty of the emissions. A Monte Carlo analysis was
 336 conducted as less than 30 measurement and estimation values were available in the
 337 present study (Maurice et al., 2000). Monte Carlo simulation gives the probability
 338 distributions of output data. In this analysis, a random emission value from an emission
 339 range is selected for a distribution model in each run or iteration. Monte Carlo
 340 simulation calculates the model a programmed number of times, each time using
 341 different randomly-selected values. Emissions are independent random variables and
 342 therefore lognormal distribution was used for the emission model, as emission

343 parameters were non-negative values and standard deviation was greater than 30 %
344 (Ramírez et al., 2008). The lognormal distribution is the probability distribution where
345 the natural logarithm of the observed values are normally distributed (Weidema et al.,
346 2013). The software tool SimaPro version 8.3.0 was used to carry out the Monte Carlo
347 simulation. In this analysis, the entire simulation was run 10,000 times for each gas
348 emission and for the emission comparisons (Meyer, 2007; Sonnemann et al., 2003). The
349 confidence interval was 95 %. The comparison of measurements and estimates for a
350 certain gas was done running the simulation of both total emissions at the same time. In
351 this case, the probability distribution function was for the difference between measured
352 and estimated values.

353

354 **3. Results**

355 Results obtained from manure sampling, field emission measurements, and from Tier
356 estimations were useful to characterise the emissions produced at the biogas plant.
357 Calculated estimates from the different IPCC approaches were assessed and their
358 respective results compared. Finally, results obtained from field measurements were
359 compared with the estimates. Results are presented in Tables 2 and 3, and in Figures 1
360 and 2.

361

362 **3.1 Biogas plant characterization**

363 Results for the characterization of the biogas plant were from the analysis of sampled
364 materials and emission measurements at the biogas plant, and they are detailed in the
365 following sections.

366

367 3.1.1 Biogas plant performance

368 As can be seen in Table 2, between the influent and the effluent of the anaerobic
369 digesters (AD2 sample, namely digestate) there is a mass flow reduction of TS, VS and
370 COD, as a result of organic matter degradation and biogas production. The slight
371 decrease in TKN between these two samples can be attributed to the NH_3 released
372 together with the biogas. Oppositely, ammonia nitrogen showed a 21 % increase, due to
373 organic nitrogen ammonification.

374 Mechanical separation produced two fractions with different nutrient contents. The
375 liquid fraction contained 95.4 % of TKN and 77.6 % of VS in the original non-separated
376 digestate. Similarly, 95.6 % of the $\text{NH}_4\text{-N}$ in the original digestate remained in the
377 liquid fraction, the same separation efficiency as for TKN. The solid fraction was rich in
378 organic matter (238.9 g L^{-1} VS) and phosphorus (2.28 g L^{-1} P) and the liquid fraction
379 was rich in soluble nitrogen (2.45 g L^{-1} $\text{NH}_4\text{-N}$) and potassium (1.20 g L^{-1} K). The
380 TAN/TKN ratio was similar in both the liquid and solid fractions of the digestate, 0.65
381 and 0.62 respectively, but in the final storage pond it increased up to 0.72, probably due
382 to an ammonification process occurred during the long period of storage. Contrarily, the
383 TAN/TKN ratio was much lower in the final compost (0.20) – biomass growth during
384 the composting process fixing ammonia, high temperatures reached during the process
385 favouring ammonia emissions, and the nitrification process, explain these differences in
386 the ratio. These processes were confirmed by the lower amount of $\text{NH}_4\text{-N}$ in the final
387 storage pond and compost piles, with respect to the liquid and the solid fraction,
388 respectively. During the liquid fraction storage, apart from the aforementioned
389 ammonification process, a basic pH of 8 favoured the volatilization of NH_3 . From these
390 results, total nitrogen losses amounted to 75.3 t y^{-1} , most of them attributable to

391 ammonia emissions. Nevertheless, sedimentation of solids and the non-
392 representativeness of the samples are issues that should be taken into account when
393 estimating emissions with the mass balance method.
394 .

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395 **Table 2.** Mass and component balance of manure samples at the biogas plant. Annual
 396 average values and standard deviation. IS: influent storage; AD1' and AD1'': Primary
 397 anaerobic digestors; AD2: Secondary anaerobic digestor; LF: liquid fraction; SF: Solid
 398 fraction; LFS: liquid fraction storage; SFS: solid fraction storage; EC: Electrical
 399 conductivity; TS: Total solids; VS: Volatile solids; COD: Chemical oxygen demand;
 400 TKN: Total Kjehldahl nitrogen; NH₄-N: Ammonium nitrogen; P: Phosphorus; K:
 401 Potassium.

Component	IS	AD1'	AD1''	AD2	LF	SF	LFS	SFS
Flow (t y ⁻¹)	38,325 ±1,864	17,337 ±63	17,340 ±3	34,683 ±137	33,888 ±235	989 ±245	33,888 ±235	396 ±98
pH	6.4 ±0.6	8.0 ±0.2	7.9 ±0.2	8.0 ±0.2	8.1 ±0.2	8.5 ±0.5	8.1 ±0.3	8.0 ±0.6
EC (mS)	7.8 ±4.3	- -	- -	- -	10.8 ±5.3	2.8 ±3.1	8.7 ±4.5	2.9 ±3.5
TS (t y ⁻¹)	3,382 ±779	780 ±150	784 ±160	1,570 ±372	1,225 ±243	279 ±73	798 ±454	125 ±46
VS (t y ⁻¹)	2,911 ±595	553 ±113	554 ±122	1,100 ±284	821 ±174	237 ±61	520 ±329	104 ±37
COD (t y ⁻¹)	5,911 ±1,702	965 ±200	948 ±184	1,829 ±357	- -	- -	- -	- -
TKN (t y ⁻¹)	153 ±30	69 ±7.6	69 ±8.0	140 ±19	129 ±13	6.3 ±1.8	73.8 ±18.0	3.6 ±1.7
NH ₄ -N (t y ⁻¹)	73.8 ±10	43.9 ±6.2	43.1 ±3.9	89.2 ±8.4	84.0 ±9.3	3.9 ±2.0	53.0 ±11	0.7 ±0.4
P (t y ⁻¹)	27.3 ±6.5	- -	- -	- -	21.0 -	2.2 ±0.9	12.9 ±8.0	1.0 ±0.4
K (t y ⁻¹)	42.7 ±15.7	- -	- -	- -	40.7 ±12.1	0.7 ±0.5	34.2 ±15.7	0.3 ±0.2

402

403 3.1.2 Emission measurement

404 Emission rates per each type of gas and for each type of storage compartment are
405 presented in Table 3. Per type of storage compartment, the highest emission factors (mg
406 $\text{m}^{-2} \text{s}^{-1}$) were for CH_4 in IS and for $\text{NH}_3\text{-N}$ in LFS and SFS. The CH_4 emission factor in
407 IS was 4.2-fold higher than in the LFS, as CH_4 production is much higher before
408 anaerobic digestion. Even though the capacity to produce CH_4 was reduced after
409 anaerobic digestion, the highest amount of CH_4 emissions per year was produced in
410 LFS. These results can be explained by the fact that storage dimensions and storage
411 time were higher for the liquid fraction (98.7 % of the total storage time) than for the
412 influent (1.3 % of the total storage time). Consequently, the loss of methane per VS
413 content was higher in LFS (15.3 %) than in IS (0.36 %). $\text{NH}_3\text{-N}$ showed the highest
414 emission factor in LFS ($0.356 \text{ mg m}^{-2} \text{ s}^{-1}$) and, taking into account the loss of $\text{NH}_3\text{-N}$ per
415 nitrogen content, it rose up to 28.6 % per TKN content, and to 39.9 % per TAN content.
416 The highest $\text{N}_2\text{O-N}$ emission factor was in SFS ($0.003 \text{ mg m}^{-2} \text{ s}^{-1}$) also taking into
417 account the amount of N_2O produced per TKN content (0.98 %). The composting
418 process in aerobic conditions favoured the production of N_2O , whilst the production of
419 N_2O per TKN content was negligible in IS and LFS (0.00 % and 0.26 %, respectively).
420 For H_2S , the highest emission factor was found in IS ($0.027 \text{ mg m}^{-2} \text{ s}^{-1}$) showing similar
421 values in LFS and in SFS (0.019 and $0.020 \text{ mg m}^{-2} \text{ s}^{-1}$, respectively).

422
423 **Table 3.** Gas emission rates (mean and standard deviation, $\text{mg m}^{-2} \text{ s}^{-1}$) of field
424 measurements per point of storage: influent storage (IS), liquid fraction storage (LFS)
425 and solid fraction storage (SFS).

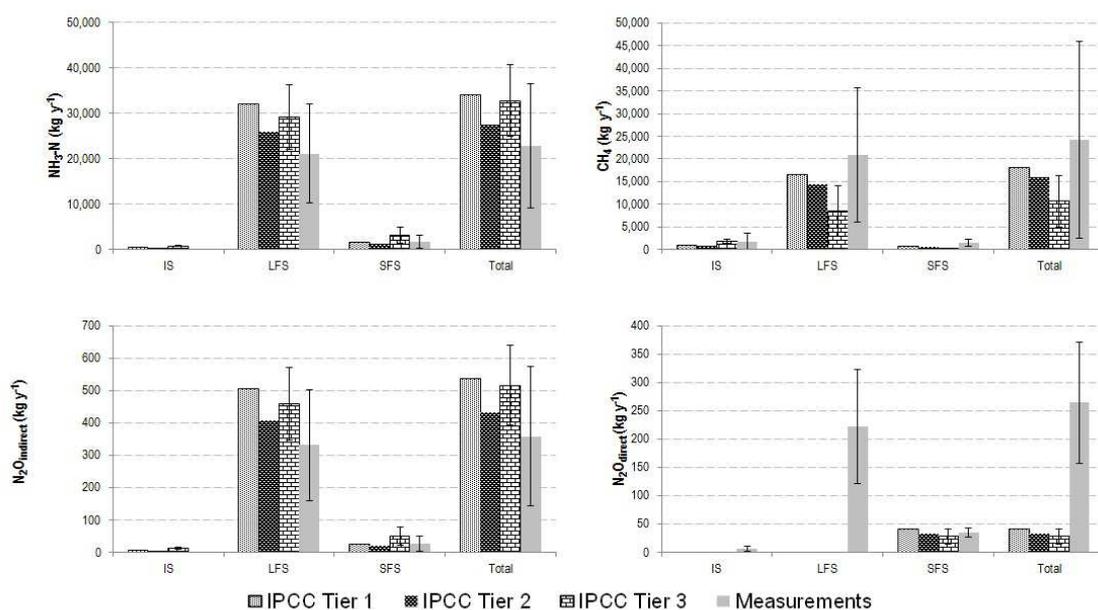
426

Storage	NH ₃ -N mg m ⁻² s ⁻¹	CH ₄ mg m ⁻² s ⁻¹	N ₂ O-N mg m ⁻² s ⁻¹	H ₂ S mg m ⁻² s ⁻¹
IS	0.032±0.021	0.983±0.887	0.002±0.001	0.027±0.014
LFS	0.356±0.172	0.235±0.233	0.002±0.001	0.019±0.013
SFS	0.152±0.164	0.136±0.096	0.003±0.001	0.020±0.021

427

428 The highest emissions for all the measured gases were produced in the storage
429 compartment of the liquid fraction of the digestate at the final storage pond (Fig. 1). An
430 emission contribution in a compartment was the ratio between the amount of this
431 emission divided by the total amount of this gas in the biogas plant, in percentage (%).
432 Liquid fraction contributions to the total plant measured emissions were 92.5 % for
433 NH₃-N, 86.4 % for CH₄, 92.5 % for N₂O_{indirect}, 84.1 % for N₂O_{direct}, and 72.6 % for H₂S
434 emissions. These high emission values found at the final storage pond were due to the
435 fact that, after mechanical separation of the digestate, the majority of N and VS content
436 remained in the liquid fraction, and also because the emitting surface of final storage
437 pond was 34-fold larger than influent storage pond emitting surface. Emissions at IS
438 were 0.2 % of NH₃-N and N₂O_{indirect}, 7.5 % of CH₄, 2.6 % of N₂O_{direct}, and 7.3 % of
439 H₂S; and emissions at SFS were 7.2 % of NH₃-N and N₂O_{indirect}, 6.1 % of CH₄, 13.3 %
440 of N₂O_{direct}, and 20.1 % of H₂S (Table S2 in supplementary information).

441



442

443 Fig. 1. NH₃-N, CH₄, N₂O_{indirect} and N₂O_{direct} emission values for IPCC models and field
 444 measurements at the influent storage (IS), the liquid fraction storage (LFS) and the solid
 445 fraction storage (SFS), mean value and standard deviation for Tier 3 and field
 446 measurement comparison.

447

448 3.2 Emission estimates adapting the IPCC guidelines

449 For all gases, the highest total estimates were those calculated with the Tier 1 approach,
 450 followed by Tier 3 and Tier 2 for NH₃-N and N₂O_{indirect} emissions, and by Tier 2 and
 451 Tier 3 for N₂O_{direct} and CH₄ emissions. NH₃-N and N₂O_{indirect} Tier 1 results were 4 %
 452 and 24 % higher than Tier 3 and Tier 2 results, respectively. N₂O_{direct} emissions were 24
 453 % higher than Tier 2 and 46 % higher than Tier 3 estimates. Methane emissions in Tier
 454 1 were 15 % and 71 % higher than Tier 2 and Tier 3 emissions, respectively (Table S2).
 455 For each gas and type of storage compartment, Tier 1 emissions were higher than Tier 2
 456 emissions. The emission contribution of the different sampling points was the same in
 457 both Tiers and for all gases. This was because the considered emission factors were

458 always the same, the only difference between Tier 1 and Tier 2 being the initial amount
459 of N and VS to calculate emissions.

460 From IPCC estimations, the results for each different gas emission showed that the
461 storage compartment of the liquid fraction of the digestate produced the highest amount
462 of emissions in the plant, results similar to those obtained by field measurements.

463

464 **3.3 Methods assessment**

465 In this section, the results from the different field measurements were compared with
466 the estimates. This comparison showed that total emissions at the biogas plant were all
467 in the same order of magnitude for $\text{NH}_3\text{-N}$, $\text{N}_2\text{O}_{\text{indirect}}$ and CH_4 gases (Fig. 1).

468 Differences were found when comparing field measurements with estimated total
469 values, being $\text{NH}_3\text{-N}$ and $\text{N}_2\text{O}_{\text{indirect}}$ measurements lower than any of the estimates and
470 CH_4 measurements higher than any of the estimates. $\text{NH}_3\text{-N}$ and $\text{N}_2\text{O}_{\text{indirect}}$ field
471 measured emissions were 33 %, 17 % and 30 % lower than Tier 1, Tier 2 and Tier 3
472 estimates, respectively. In IPCC approach, the amount of NH_3 and N_2O emissions was
473 estimated according to the N content in excreta. The higher N content in Tier 1 and Tier
474 3 than in Tier 2 delivered higher N emissions in Tier 1 and Tier 3, and consequently,
475 higher differences of N emissions comparing with measurements. Total CH_4 field
476 measured emissions were 33 %, 52 % and 2.27-fold higher than Tier 1, Tier 2 and Tier
477 3 estimates, respectively (Table S3).

478 Other differences could be observed when comparing emissions per type of storage
479 compartment. $\text{NH}_3\text{-N}$ and $\text{N}_2\text{O}_{\text{indirect}}$ measurements were 53 % lower than Tier 3
480 estimates, and 6 % and 31 % higher than Tier 1 and Tier 2 estimates, in the SFS. Still,
481 field measurements in the IS were between 86 % and 94 % lower than the estimates. In

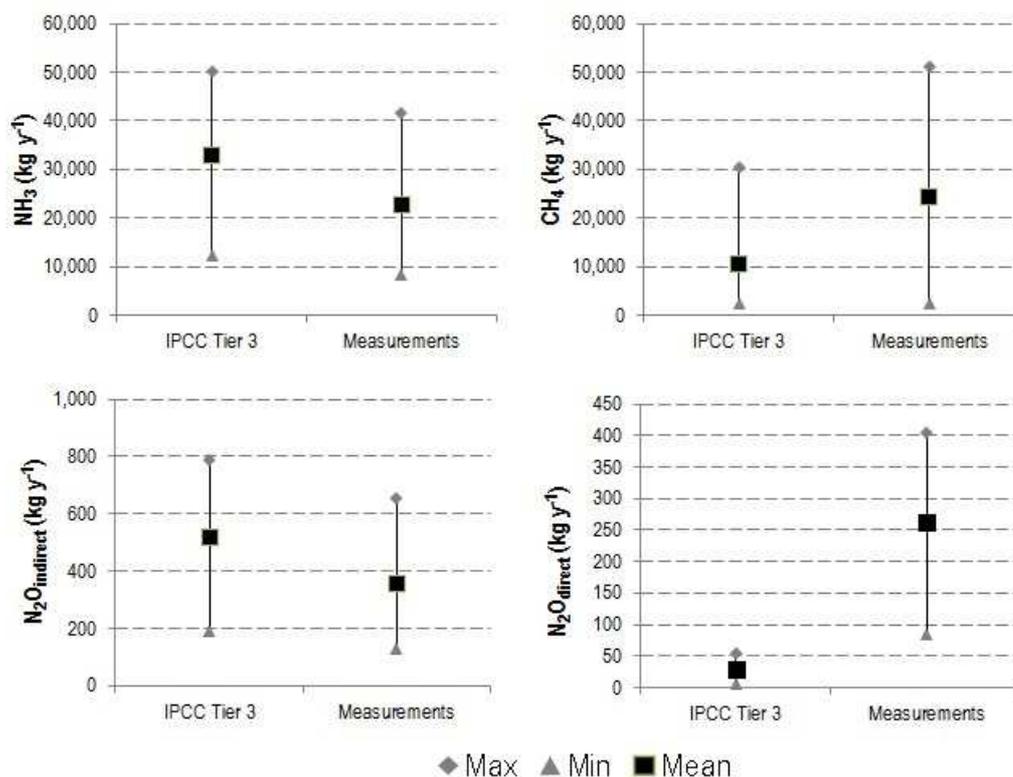
482 this case, even though these differences were so high, they did not exert much impact
483 because the contribution of IS emissions to the total emissions was very small (Table
484 S2).

485 On the other hand, ratios between measurement and Tier 1, Tier 2 and Tier 3 results
486 were nearly the same at LFS (0.66, 0.81 and 0.72) and total values (0.67, 0.83 and 0.70).
487 This can be explained because the contribution of LFS to total emissions was very high
488 and with similar percentages in Tier estimates and field measurements, being 94.1 % in
489 Tier 1 and Tier 2, 88.9 % in Tier 3, and 92.5 % in field measurements,.

490 In the case of CH₄ emissions, measurements at LFS were 1.27-fold, 1.45-fold and 2.47-
491 fold higher than Tier 1, Tier 2 and Tier 3 estimates, respectively, similar values than
492 when comparing total values (1.33-fold, 1.52-fold and 2.27-fold, respectively) (Table
493 S3). Measured emissions were 5.31-fold higher than Tier 3 in SFS and 5 % lower at IS.
494 High contribution of LFS to total emissions was the reason behind these results, 86.4 %
495 in field measurements and 79.5 % in Tier 3 estimates. On the other hand, Tier 3
496 estimates showed the lowest results. B₀ used to estimate Tier 3 emissions was obtained
497 from laboratory experiments, the lower methane production capacity of the digestate
498 fractions resulting in these low values.

499 The highest differences between field measurements and estimates were found in
500 N₂O_{direct} total emission results. Total N₂O_{direct} field-measured emissions were one order
501 of magnitude higher than total emission estimates in all Tiers. In this case, the most
502 significant differences were found in LFS, as N₂O_{direct} emissions of SFS were in the
503 same order of magnitude for all Tiers and field measurements. Field-measured results
504 were 6 % and 25 % higher than Tier 2 and Tier 3 estimates, respectively, and 14 %
505 lower than Tier 1 estimates.

506 A thorough analysis could be carried out with results from Tier 3 estimates and field
507 measurements as they were figured out from the compilation of multiple field data (n
508 =24). Therefore, in order to examine the differences between measurements and
509 estimates, the maximum and minimum emission values were considered. Fig. 2 shows
510 the average, maximum and minimum emissions values for all gas emissions. Tier 3 and
511 field measurement emission value ranges were similar for each gas, except for CH₄
512 maximum and N₂O emissions. NH₃-N, N₂O_{indirect}, N₂O_{direct} and CH₄ maximum values
513 were between 1.5-fold and 2.9-fold higher than the mean, and the minimum values were
514 between 2.7-fold and 10-fold lower than the mean. There were no substantial
515 differences between NH₃-N and N₂O_{indirect} means, maximum and minimum values, but
516 N₂O_{direct} emission value ranges did not match in either average, maximum or minimum
517 values. Total N₂O_{direct} field measurements emissions included emissions from IS, LFS
518 and SFS, whilst estimated Tier 3 N₂O_{direct} emissions only accounted for SFS emissions.
519 CH₄ means and maximums differed considerably, but minimum value was nearly the
520 same.
521



522

523 Fig. 2. Tier 3 estimates and field measurements, mean value with the maximum-
 524 minimum value range.

525

526 3.4 Uncertainty assessment

527 Monte Carlo simulations provided a probability distribution function of measured and
 528 Tier 3 estimated emissions, and the significance of the differences in the comparison of
 529 gas measurements with Tier 3 estimates. The results of the Monte Carlo analysis are
 530 presented in Table 4, showing the statistical parameters related to the distribution
 531 functions of NH₃-N, CH₄, N₂O and H₂S emissions and their comparisons. Results were
 532 also represented in histogram graphs showing the obtained probability distributions and
 533 are included in the supplementary information (Figures S1 to S9). The shapes of the
 534 probability distributions were different for each one of the emissions. NH₃-N, CH₄ and
 535 N₂O_{indirect} Tier 3 estimates and N₂O_{direct} measurements showed a more normal

536 distribution than $\text{NH}_3\text{-N}$, CH_4 $\text{N}_2\text{O}_{\text{indirect}}$ and H_2S measurements and $\text{N}_2\text{O}_{\text{direct}}$ Tier 3
537 estimates.

538 The coefficient of variation (CV) is the ratio between the standard deviation and the
539 mean, and indicates the dispersion of the measures around the mean. $\text{NH}_3\text{-N}$, N_2O and
540 estimated CH_4 CVs were between 8.1 % and 35.3 %. CH_4 and H_2S CVs of
541 measurement were higher than 200 %, showing the highest uncertainties.

542

543 **Table 4.** Uncertainty analysis of total NH₃- N, CH₄, N₂O_{indirect} and N₂O_{direct} measured and Tier 3 estimated emissions, using Monte Carlo
 544 simulation for total emission value at the biogas plant. Confidence interval: 95%. Included parameters are: Mean, median, SD (Standard
 545 Deviation), CV (Coefficient of variation), confidence interval limit 2.5%, confidence interval limit 95.7% and SEM (standard error of the
 546 mean

Emissions		Unit	Mean	Median	SD	CV (%)	2.5%	97.5%	SEM
<i>Emission analysis</i>									
NH ₃ - N	measurement	kg	2.3E+04	2.1E+04	8.1E+03	35.3	1.1E+04	4.2E+04	80.5
	Tier 3 IPCC	kg	3.3E+04	3.3E+04	2.6E+03	8.1	2.8E+04	3.8E+04	26.4
CH ₄	measurement	kg	2.4E+04	6.1E+03	7.8E+04	329.6	2.4E+02	1.5E+05	780.4
	Tier 3 IPCC	kg	1.1E+04	1.0E+04	2.5E+03	23.5	6.5E+03	1.6E+04	25.0
N ₂ O _{indirect}	measurement	kg	3.6E+02	3.4E+02	1.3E+02	35.7	1.7E+02	6.7E+02	1.3
	Tier 3 IPCC	kg	5.0E+02	5.0E+02	4.1E+01	8.1	4.2E+02	5.8E+02	0.4
N ₂ O _{direct}	measurement	kg	2.6E+02	2.6E+02	5.0E+01	18.9	1.8E+02	3.7E+02	0.5
	Tier 3 IPCC	kg	2.7E+01	2.6E+01	7.7E+00	28.5	1.5E+01	4.5E+01	0.1
H ₂ S	measurement	kg	8.5E+02	3.5E+02	2.0E+03	239.5	2.5E+01	4.7E+03	20.3
		% measure ≥ estimate							
<i>Emission comparison</i>									
NH ₃ - N	measurement - estimate	12.2	-1.0E+04	-1.1E+04	8.4E+03	-84	-2.3E+04	1.0E+04	83.8
	measurement - estimate	37.8	1.4E+04	-4.1E+03	1.1E+05	774	-1.4E+04	1.5E+05	1075.3
CH ₄	measurement - estimate	13.7	-1.4E+02	-1.6E+02	1.3E+02	-96	-3.5E+02	1.8E+02	1.3
	measurement - estimate	100.0	2.4E+02	2.3E+02	5.0E+01	21	1.5E+02	3.5E+02	0.5
N ₂ O _{indirect}	measurement - estimate								
N ₂ O _{direct}	measurement - estimate								

547

548 In the comparison of measurements with Tier 3 estimates, the probability distribution
549 function was for the difference between measurements and estimates. Results showed
550 that the probability of the runs with $\text{NH}_3\text{-N}$ measurement values higher or equal than
551 estimate values was 12.2 %, similar to the 13.7 % probability of $\text{N}_2\text{O}_{\text{indirect}}$. The
552 probability of measurement values higher or equal than estimate values was 37.8 % for
553 CH_4 and 100 % $\text{N}_2\text{O}_{\text{direct}}$ emissions. In this comparative analysis, CH_4 differences also
554 showed the highest CV.

555

556 **4. Discussion**

557 Emissions producing relevant environmental impact from a biogas plant were selected.
558 Field measurements and the modified IPCC estimation approaches used in the study
559 served to characterise gas emissions. To calculate emissions of anaerobic digestion by-
560 products, different hypothesis based on literature and laboratory results were
561 established. These methods can be useful for preparing an emission inventory.
562 Nevertheless, each method shows advantages and drawbacks that should be taken into
563 account in their application. Results to characterise the biogas plant and the methods
564 used are discussed in this section.

565 Based on the four emission inventories the biogas plant under study could be assessed.
566 Even emission values showed high differences among the different approaches used,
567 total emissions were all in the same order of magnitude for $\text{NH}_3\text{-N}$, $\text{N}_2\text{O}_{\text{indirect}}$ and CH_4
568 gases. The storage of liquid fraction of digestate made the highest emissions in the
569 plant. Therefore, actions should focus in digestate phase in order to reduce emissions
570 and environmental impacts. Anaerobic digestion can be used in combination with other
571 processing technologies. Some of them are more effective when performed after

572 anaerobic digestion, such as evaporation (Bonmatí et al., 2003; Bonmatí and Flotats,
573 2003b) or ammonia removal by air stripping (Bonmatí and Flotats, 2003a). Moreover,
574 anaerobic digestion can be integrated with struvite precipitation in order to remove
575 ammonia and phosphates from manure, producing a valuable fertilizer in the meantime
576 (Cerrillo et al., 2015), and also combined with a composting process of the solid
577 fraction of the digestate (Bustamante et al., 2014; Bustamante et al., 2013). Biogas
578 production is also a clean energy source when processing other organic wastes such as
579 the organic fraction of municipal solid waste or sewage sludge from wastewater
580 treatment plants (Silvestre et al., 2011).

581 Regarding field emission measurements sampling representativeness is a critical point,
582 due to the high number of samples needed to cover the high variability in conditions. In
583 the present case, the two-year period could be long, but with the eight sampling
584 campaigns carried out the representativeness of the samples could not be assured for all
585 the different climates and plant operation conditions throughout an entire year. On the
586 other hand, emission sampling is time-consuming, requires technical knowledge, is not
587 always affordable, and could over- or under- estimate emissions; e.g. Lindvall Hood
588 method is said to overestimate emissions because local condition are modified during
589 sampling. This is where emission models come as an interesting tool to determine
590 emission rates when field measurements are not possible. Nevertheless, it would be
591 advisable to test the different models in local conditions, and to compare them with field
592 measurements in order to establish their accuracy. In order to validate these
593 methodologies, more biogas plants should be analysed.

594 The IPCC guidelines provided different approaches to determine CH₄, NH₃ and N₂O
595 emissions based on the amount of VS produced by cows, in the case of CH₄ emissions,

596 and on the amount of nitrogen excreted by cows in the case of NH_3 and N_2O emissions.
597 Emission factors were provided for different manure management systems, but there
598 were no available factors either for the digestate, for the solid fraction or the liquid
599 fraction, after the solid-liquid separation of digestate. Therefore, it was necessary to
600 establish a hypothesis and, based on the existing literature, select emission factors and
601 additional considerations such as the distribution of N and VS depending on mechanical
602 separation efficiencies and the decrease in VS after anaerobic digestion (see section
603 2.4.1 to 2.4.3), to estimate the emissions for the scenario under study. All these
604 considerations could be used for further studies.

605 The selection of emission factors was a critical point. For ammonia emissions, many
606 studies were found providing data for emissions produced by the digestate storage
607 compartment. NH_3 emissions from uncovered digested slurry were twice as those from
608 untreated dairy cattle slurry, due to the high $\text{NH}_4\text{-N}$ content and the pH value of
609 anaerobically digested slurry (Amon et al., 2006; Clemens et al., 2006; Maurer and
610 Müller, 2012). Few studies were found for cattle emissions of the digestate fractions
611 after liquid-solid separation (Holly et al., 2017; Perazzolo et al., 2015). IPCC NH_3
612 estimated emission factors were 40 % and 30 % of the total N content of the slurry in
613 LFS and SFS, respectively. Considering field-measured emissions and TKN amount at
614 the final storage pond, the estimated NH_3 emission factor rose to 28.6 %; and to 42.9 %
615 when applying TKN balance between the final storage pond and the liquid fraction,
616 which was a little higher than the IPCC NH_3 emission factor (40 %). In view of the
617 results of this study and the factors reported in literature, $\text{NH}_3\text{-N}$ emission factor
618 variability for manure digestate could be observed. Therefore, more biogas plants

619 should be analysed to determine specific emission factors for manure after anaerobic
620 digestion.

621 On the other hand, annual $\text{NH}_3\text{-N}$ emissions obtained by field measurements were lower
622 than in any of the IPCC estimates. This low value could be because not all $\text{NH}_3\text{-N}$
623 emissions at the plant were measured, since there might be losses at the plant which
624 were not accounted for (such as losses during the mechanical liquid-solid separation of
625 the digestate). The high $\text{NH}_3\text{-N}$ emissions value obtained with the mass balance method,
626 confirm this hypothesis.

627 When it comes to CH_4 emissions estimation, a literature review was conducted to search
628 for the maximum methane producing capacities (B_0) of the different digestate fractions,
629 and MCFs for Tier 1 and Tier 2 calculations. B_0 may be influenced by the proportion
630 and type of manure and co-substrate, and the HRT of the reactor (Amon et al., 2006;
631 Gioelli et al., 2011; Lehtomäki et al., 2007; Rico et al., 2011; Thygessen et al., 2014) .
632 A B_0 of $0.121 \text{ m}^3 \text{ CH}_4 \cdot \text{kg VS}^{-1}$ was reported by Rodhe et al. (2015) in a digestate from
633 the anaerobic co-digestion of cattle manure (95 %) and food waste and solid cattle
634 manure (5 %). This was the selected value, it being obtained in a similar scenario to the
635 study in case. Nevertheless, CH_4 estimated emissions in Tier 1 and Tier 2 might have
636 been overestimated, as the factor used was obtained from the non-separated digestate
637 storage compartment.

638 For Tier 3, and in order to more accurately estimate methane emissions, batch tests of
639 the solid and liquid fractions of the digestate were performed (see section 2.3). The
640 methane potential capacities obtained were much lower than the reported results in
641 literature, maybe due to the different feedstock composition, the climate conditions in
642 the area and the long HRT. From field measurement emissions and the VS content of

643 manure, the calculated methane production capacities were $0.214 \text{ m}^3 \text{ CH}_4 \cdot \text{kg VS}^{-1}$ for
644 LFS and $0.533 \text{ CH}_4 \cdot \text{m}^3 \text{ kg VS}^{-1}$ for SFS. These values are much higher than those
645 obtained from the batch tests, indicating a discrepancy in the methane potential VS
646 capacity, from experimental to calculation methods. In view of this variability in results
647 from tests, calculations and literature, more studies to harmonize the methane potential
648 capacity in different types of manure would be convenient.

649 Total field measurement methane emissions were higher than estimates. For instance,
650 the difference between field-measured total value and Tier 3 total estimate could be
651 explained by the low methane capacity of VS in the digestate samples obtained for the
652 batch tests. Moreover, there arose the possibility that a certain quantity of CH_4 gas
653 could have been retained in the digestate after anaerobic digestion, its later release in
654 storage producing higher emissions. It has been described that up to 20 % of biogas
655 could be released in the digestate storage compartment (DEA, 1995). According to the
656 results in this study, it seems that the adapted IPCC models underestimate potential CH_4
657 emissions. These findings are similar to those reported by Owen and Silver (2015), who
658 compared greenhouse emissions from dairies of thirty-eight studies with Tier 2
659 modelled values. The study concluded that current greenhouse gas emission factors
660 generally underestimate emissions from dairy manure and that more field data are
661 needed to refine models. It has been found that measurements were higher than Tier 1,
662 Tier 2 and Tier 3 estimates. In Tier 3 estimations, MCFs were selected according to the
663 average annual temperature, being the same for all Tiers. Difference in results were
664 mainly due to the different amount of VS used to estimate emissions. Even though the
665 initial amount of VS was the highest in Tier 3, total yearly emissions were the lowest of
666 the three Tiers. Since most of CH_4 emissions were produced in LFS, a lower VS content

667 in LFS from analytical data used to estimate in Tier 3 provided fewer emissions in Tier
668 3 than in Tier 1 and Tier 2.

669

670 Authors reported that N_2O_{direct} estimated emissions were less than half compared to
671 field-measured emissions in anaerobic lagoons and piles, but greater than in the slurry
672 storage compartment. Contrarily, N_2O_{direct} measurements were 9.4-fold higher than Tier
673 3 estimates in the present study. This could be explained by the fact that the N_2O_{direct}
674 emission factor at the liquid slurry storage compartment was considered to be zero,
675 according to IPCC guidelines, even if it was found to be present in the samples taken.

676 As for H_2S emissions, results showed that emission rates were higher in IS (pH 6.4)
677 than in LFS (pH 8.1) and SFS (pH 8.0) (Table 3). These results are in concordance with
678 results from Andriamanohiarisoamanana et al. (2015) and Sutaryo et al. (2012), who
679 concluded that pH has a great influence on sulphide emission rates from dairy manure.

680 A Monte Carlo analysis was carried out to examine the uncertainties of emissions. This
681 method has been used for the statistical analysis of emissions released in electricity
682 production studies (Fantin et al., 2015; Maurice et al., 2000; Sonnemann et al., 2003), or
683 in emission inventories (Ramírez et al., 2008). CH_4 and H_2S total emissions showed the
684 highest coefficients of variation. On the other hand. The comparison of emission
685 distribution functions delivered that the probability of Tier 3 estimates to be higher than
686 measurements was 88 % for NH_3-N , 86 % for N_2O_{indirect} , 62 % for CH_4 and null for
687 N_2O_{direct} total emission values.

688 A reference situation was defined to distinguish the current manure management
689 practices from the situation previous to the implementation of the biogas treatment plant
690 (see Section 3 of Supplementary information). Comparing the current situation with the

691 reference situation, CH₄, NH₃-N and indirect N₂O total emissions were lower in the
692 current situation, demonstrating the potential benefits of manure processing at a biogas
693 plant (Table S6 in supplementary information). The highest reductions were for CH₄.
694 Comparing total CH₄ measurements in the current situation with Tier 1, Tier 2 and Tier
695 3 estimates in the reference situation, reductions were 69 %, 64 % and 80 %, respectively.
696
697 The characterisation of a biogas plant by field measurements is a complex task and not
698 always possible. The alternative is model estimation according to the specific situation.
699 In this sense, and from the results of this study, emission estimations following the
700 adapted approaches for a biogas plant proposed in this study could be used to determine
701 by-product emissions from anaerobic digestion in similar scenarios. Emission follow-up
702 in this type of manure management system could be useful for decision making, in order
703 to mitigate emissions and contribute with cleaner manure management practices. In this
704 sense, emissions should be estimated with as much precision as possible to attain
705 accurate results. Therefore, further research could focus on determining specific
706 emission factors in different biogas plants to achieve this goal.

707

708 **6. Conclusions**

709 Emissions at an anaerobic co-digestion plant were characterised by field measurements
710 and IPCC estimation approaches. IPCC guidelines were adapted using emission factors
711 and several considerations for the specific conditions. In Tier 1 and Tier 2, data from
712 literature were used. In Tier 3, analytical data were used. Comparing measurements
713 with estimations, results were found to be in the same order of magnitude. However,
714 more accurate emission factors from field measurements are needed, which might be

715 useful for preparing gaseous emission inventories for waste management systems
716 environmental assessments. On the other hand, biogas plant can be considered an
717 efficient solution to reduce emissions from traditional manure storage, as $\text{NH}_3\text{-N}$, CH_4
718 and $\text{N}_2\text{O}_{\text{indirect}}$ emissions from by-products storage after anaerobic digestion were lower
719 than those from manure stored in an uncovered pond in a reference situation.

720

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726

727 **References**

- 728 Aguirre-Villegas, H.A., Larson, R.A., 2017. Evaluating greenhouse gas emissions from
729 dairy manure management practices using survey data and lifecycle tools. *J. Clean*
730 *Prod.* 143, 169-179.
- 731 Amon, B., Kryvoruchko, V., Amon, T., Zechmeister-Boltenstern, S., 2006. Methane,
732 nitrous oxide and ammonia emissions during storage and after application of dairy
733 cattle slurry and influence of slurry treatment. *Agr. Ecosyst. Environ.* 112, 153-162.
- 734 Andriamanohiarisoamanana, F.J., Sakamoto, Y., Yamashiro, T., Yasui, S., Iwasaki, M.,
735 Ihara, I., Tsuji, O., Umetsu, K., 2015. Effects of handling parameters on hydrogen
736 sulfide emission from stored dairy manure. *J. Environ. Manage.* 154, 110-116.

- 737 APHA, 2014. Standard Methods for the Examination of Water and Wastewater. 22nd
738 ed. APHA (American Public Health Association) , AWWA (American Public
739 Health Association) , WEP (Water Environment Federation), Whashington, DC.
- 740 Bonmatí, A., Campos, E., Flotats, X., 2003. Concentration of pig slurry by evaporation:
741 anaerobic digestion as the key process. *Water Sci. Technol.* 48, 189-194.
- 742 Bonmatí, A., Flotats, X., 2003a. Air stripping of ammonia from pig slurry:
743 characterisation and feasibility as a pre- or post-treatment to mesophilic anaerobic
744 digestion. *Waste Manage.* 23, 261-272.
- 745 Bonmatí, A., Flotats, X., 2003b. Pig slurry concentration by vacuum evaporation:
746 Influence of previous mesophilic anaerobic digestion process. *J. Air Waste Manage.*
747 *Assoc.* 53, 21-31.
- 748 Braschel, N., Posch, A., 2013. A review of system boundaries of GHG emission
749 inventories in waste management. *J. Clean Prod.* 44, 30-38.
- 750 Bustamante, M.A., Moral, R., Bonmatí, A., Palatsi, J., Solé-Mauri, F., Bernal, M.P.,
751 2014. Integrated Waste Management Combining Anaerobic and Aerobic Treatment:
752 A Case Study. *Waste Biomass Valorization* 5, 481-490.
- 753 Bustamante, M.A., Restrepo, A.P., Albuquerque, J.A., Pérez-Murcia, M.D., Paredes,
754 C., Moral, R., Bernal, M.P., 2013. Recycling of anaerobic digestates by composting:
755 effect of the bulking agent used. *J. Clean Prod.* 47, 61-69.
- 756 Cerrillo, M., Palatsi, J., Comas, J., Vicens, J., Bonmati, A., 2015. Struvite precipitation
757 as a technology to be integrated in a manure anaerobic digestion treatment plant –
758 removal efficiency, crystal characterization and agricultural assessment. *J. Chem.*
759 *Technol. Biotechnol.* 90, 1135-1143.

- 760 Cherubini, E., Zanghelini, G.M., Alvarenga, R.A.F., Franco, D., Soares, S.R., 2015.
761 Life cycle assessment of swine production in Brazil: a comparison of four manure
762 management systems. *J. Clean Prod.* 87, 68-77.
- 763 Clemens, J., Trimborn, M., Weiland, P., Amon, B., 2006. Mitigation of greenhouse gas
764 emissions by anaerobic digestion of cattle slurry. *Agr. Ecosyst. Environ.* 112, 171-
765 177.
- 766 DEA, 1995. Danish Energy Agency. Progress report on the economy of centralized
767 biogas plants, in: BioPress (Ed.), Copenhagen, p. 34.
- 768 Dinuccio, E., Berg, W., Balsari, P., 2008. Gaseous emissions from the storage of
769 untreated slurries and the fractions obtained after mechanical separation. *Atmos.*
770 *Environ.* 42, 2448-2459.
- 771 EEA, 2013. EMEP/EEA air pollutant emission inventory guidebook - 2013, update July
772 2015.
- 773 EEC, 1991. Council Directive of 12 December 1991 concerning the protection of waters
774 against pollution caused by nitrates from agricultural sources (91/676 /EEC). *Off. J.*
775 *Eur. Union.* 91/67/EEC, L375/371-378.
- 776 EEC, 2001. Directive 2001/81/EC of the European Parliament and of the Council of 23
777 October 2001 on national emission ceilings for certain atmospheric pollutants. *Off.*
778 *J. Eur. Union.* 2001/81/EC, L 309/322-330.
- 779 EEC, 2009. Directive 2009/28/EC of the European Parliament and of the Council of 23
780 April 2009 on the promotion of the use of energy from renewable sources and
781 amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. *Off.*
782 *J. Eur. Union.* 2009/28/EC, L 140/116-162.

- 783 El-Mashad, H.M., Zhang, R., 2010. Biogas production from co-digestion of dairy
784 manure and food waste. *Bioresource Technol.* 101, 4021-4028.
- 785 EPA, 2001. Environmental Protection Agency. Odour Impacts and Odour Emission
786 Control Measures for Intensive Agriculture. Final report. Environmental Research.
787 R&D Report Series No. 14, Johnstown Castle Estate, Wexford, Ireland. ISBN: 1-
788 84095-075-7.
- 789 EPA draft, 1997. Procedure For Collection And Analysis Of Ammonia In Stationary
790 Sources Conditional Test Method (CTM-027).
- 791 Fantin, V., Giuliano, A., Manfredi, M., Ottaviano, G., 2015. Environmental assessment
792 of electricity generation from an Italian anaerobic digestion plant. *Biomass.*
793 *Bioenerg.* 83, 422-435.
- 794 Flotats, X., Foged, H.L., Bonmati, A., Palatsi, J., Magri, A., Schelde, K.M., 2011.
795 Manure Processing technologies. Technical Report No. II concerning “Manure
796 Processing Activities in Europe” to the European Commission, Directorate-General
797 Environment. Project reference: ENV.B.1/ETU/2010/0007.
798 <<http://hdl.handle.net/2117/18944>>, p. 188.
- 799 Generalitat de Catalunya, 2009. Decree 136/2009 of 1 September, approving the action
800 program applicable to vulnerable zones related to nitrate pollution coming from
801 agricultural sources and management of livestock manure. DOGC (in Catalan).
- 802 Gioelli, F., Dinuccio, E., Balsari, P., 2011. Residual biogas potential from the storage
803 tanks of non-separated digestate and digested liquid fraction. *Bioresource Technol.*
804 102, 10248-10251.

- 805 Holly, M.A., Larson, R.A., Powell, J.M., Ruark, M.D., Aguirre-Villegas, H., 2017.
806 Greenhouse gas and ammonia emissions from digested and separated dairy manure
807 during storage and after land application. *Agr. Ecosyst. Environ.* 239, 410-419.
- 808 IPCC, 2006. Guidelines for National Greenhouse Gas Inventories., in: Prepared by the
809 National Greenhouse Gas Inventories Programme, E.H.S., Buendia L., Miwa K.,
810 Ngara T. and Tanabe K. (eds). (Ed.). Intergovernmental Panel of Climate Change.
- 811 Lehtomäki, A., Huttunen, S., Rintala, J.A., 2007. Laboratory investigations on co-
812 digestion of energy crops and crop residues with cow manure for methane
813 production: Effect of crop to manure ratio. *Resour. Conserv. Recycl.* 51, 591-609.
- 814 Lijó, L., González-García, S., Bacenetti, J., Fiala, M., Feijoo, G., Moreira, M.T., 2014.
815 Assuring the sustainable production of biogas from anaerobic mono-digestion. *J.*
816 *Clean Prod.* 72, 23-34.
- 817 Loyon, L., Burton, C.H., Misselbrook, T., Webb, J., Philippe, F.X., Aguilar, M.,
818 Doreau, M., Hassouna, M., Veldkamp, T., Dourmad, J.Y., Bonmati, A., Grimm, E.,
819 Sommer, S.G., 2016. Best available technology for European livestock farms:
820 Availability, effectiveness and uptake. *J. Environ. Manage.* 166, 1-11.
- 821 Massaro, V., Digiesi, S., Mossa, G., Ranieri, L., 2015. The sustainability of anaerobic
822 digestion plants: a win-win strategy for public and private bodies. *J. Clean Prod.*
823 104, 445-459.
- 824 Maurer, C., Müller, J., 2012. Ammonia (NH₃) emissions during drying of untreated and
825 dewatered biogas digestate in a hybrid waste-heat/solar dryer. *Eng. Life Sci.* 12,
826 321-326.

- 827 Maurice, B., Frischknecht, R., Coelho-Schwartz, V., Hungerbühler, K., 2000.
828 Uncertainty analysis in life cycle inventory. Application to the production of
829 electricity with French coal power plants. *J. Clean Prod.* 8, 95-108.
- 830 Meteocat, 2016. Meteorology Service of Catalonia. Banyoles station (in Catalan).
831 <http://www.meteo.cat/>. Last consultation 27/01/2016.
- 832 Meyer, V.R., 2007. Measurement uncertainty. *Journal of Chromatography A* 1158, 15-
833 24.
- 834 Misselbrook, T. H., Pain, B. F., & Headon, D. M., 1998. Estimates of Ammonia
835 Emission from Dairy Cow Collecting Yards. *Journal of Agricultural Engineering*
836 *Research*, 71(2), 127-135.
- 837 Møller, H.B., Sommer, S.G., Ahring, B.K., 2002. Separation efficiency and particle size
838 distribution in relation to manure type and storage conditions. *Bioresource Technol.*
839 85, 189-196.
- 840 NIOSH, 1994a. The National Institute for Occupational Safety and Health. "Method
841 6013: Hydrogen sulfide," Issue 1.
- 842 NIOSH, 1994b. The National Institute for Occupational Safety and Health. "Method
843 6015: Ammonia," Issue 2.
- 844 Noguerol-Arias, J., Rodríguez-Abalde, A., Romero-Merino, E., Flotats, X., 2012.
845 Determination of Chemical Oxygen Demand in Heterogeneous Solid or Semisolid
846 Samples Using a Novel Method Combining Solid Dilutions as a Preparation Step
847 Followed by Optimized Closed Reflux and Colorimetric Measurement. *Anal. Chem.*
848 84, 5548-5555.
- 849 Owen, J.J., Silver, W.L., 2015. Greenhouse gas emissions from dairy manure
850 management: a review of field-based studies. *Global Change Biology* 21, 550-565.

- 851 Perazzolo, F., Mattachini, G., Fulvia, T., Misselbrook, T., Provolò, G., 2015. Effect of
852 mechanical separation on emissions during storage of two anaerobically codigested
853 animal slurries. *Agr. Ecosyst. Environ.* 207, 1-9.
- 854 Poeschl, M., Ward, S., Owende, P., 2012a. Environmental impacts of biogas
855 deployment – Part I: life cycle inventory for evaluation of production process
856 emissions to air. *J. Clean Prod.* 24, 168-183.
- 857 Poeschl, M., Ward, S., Owende, P., 2012b. Environmental impacts of biogas
858 deployment – Part II: life cycle assessment of multiple production and utilization
859 pathways. *J. Clean Prod.* 24, 184-201.
- 860 Ramírez, A., de Keizer, C., Van der Sluijs, J.P., Olivier, J., Brandes, L., 2008. Monte
861 Carlo analysis of uncertainties in the Netherlands greenhouse gas emission
862 inventory for 1990–2004. *Atmos. Environ.* 42, 8263-8272.
- 863 Rico, C., Rico, J.L., Tejero, I., Muñoz, N., Gómez, B., 2011. Anaerobic digestion of the
864 liquid fraction of dairy manure in pilot plant for biogas production: Residual
865 methane yield of digestate. *Waste Manage.* 31, 2167-2173.
- 866 Rodhe, L.K.K., Ascue, J., Willén, A., Vegerfors Persson, B., Nordberg, A., 2015.
867 Greenhouse gas emissions from storage and field application of anaerobically
868 digested and non-digested cattle slurry. *Agr. Ecosyst. Environ.* 199, 358-368.
- 869 Rostami, M., Monaco, S., Sacco, D., Grignani, C., Dinuccio, E., 2015. Comparison of
870 ammonia emissions from animal wastes and chemical fertilizers after application in
871 the soil. *Int J Recycl Org Waste Agricult* 4 (2): 127-134.
- 872 Silvestre, G., Rodríguez-Abalde, A., Fernández, B., Flotats, X., Bonmati, A., 2011.
873 Biomass adaptation over anaerobic co-digestion of sewage sludge and trapped
874 grease waste. *Bioresource Technol.* 102, 6830–6836.

- 875 Sommer, S.G., Petersen, S.O., Sogaard, H.T., 2000. Atmospheric pollutants and trace
876 gases: Greenhouse gas emission from stored livestock slurry. *J. Environ. Qual.* 29,
877 744-751.
- 878 Sonnemann, G.W., Schuhmacher, M., Castells, F., 2003. Uncertainty assessment by a
879 Monte Carlo simulation in a life cycle inventory of electricity produced by a waste
880 incinerator. *J. Clean Prod.* 11, 279-292.
- 881 Storm, I.M.L.D., Hellwing, A.L.F., Nielsen, N.I., Madsen, J., 2012. Methods for
882 Measuring and Estimating Methane Emission from Ruminants. *Animals* 2, 160-183.
- 883 Sutaryo, S., Ward, A.J., Møller, H.B., 2012. Thermophilic anaerobic co-digestion of
884 separated solids from acidified dairy cow manure. *Bioresource Technol.* 114, 195-
885 200.
- 886 Thygesen, O., Sommer, S.G., Shin, S.G., Triolo, J.M., 2014. Residual biochemical
887 methane potential (BMP) of concentrated digestate from full-scale biogas plants.
888 *Fuel* 132, 44-46.
- 889 Vac, S.C., Popita, G.E., Frunzeti, N., Popovici, A., 2013. Evaluation of Greenhouse Gas
890 Emission from Animal Manure Using the Closed Chamber Method for Gas Fluxes.
891 *Not. Bot. Horti Agrobot. Cluj-Na* 41, 576-581.
- 892 van Belois, H. J., Anzion, C. J. M., 1992. Measurement of emissions over surface areas
893 using the hood method. In A. J. Dragt & J. van Ham (Eds.), *Studies in*
894 *Environmental Science*, 51, 439-445
- 895 Weidema, B.P., Bauer, C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo,
896 C.O., Wernet, G., 2013. The ecoinvent database: Overview and methodology, Data
897 quality guideline for the ecoinvent database version 3, www.ecoinvent.org.
- 898

Highlights

- Emissions from biogas plant are sampled, and estimated adapting IPCC method
- Sampled and estimated NH_3 , CH_4 and N_2O emissions are in the same order of magnitude
- Specific emission factors for anaerobic digestion by-products are required
- Emissions at the biogas plant were lower than emissions in a reference situation
- Accurate emission estimation is a key point for preparing emission inventories