

1 Anaerobic digestion affecting nitrous 2 oxide and methane emissions from the 3 composting process 4

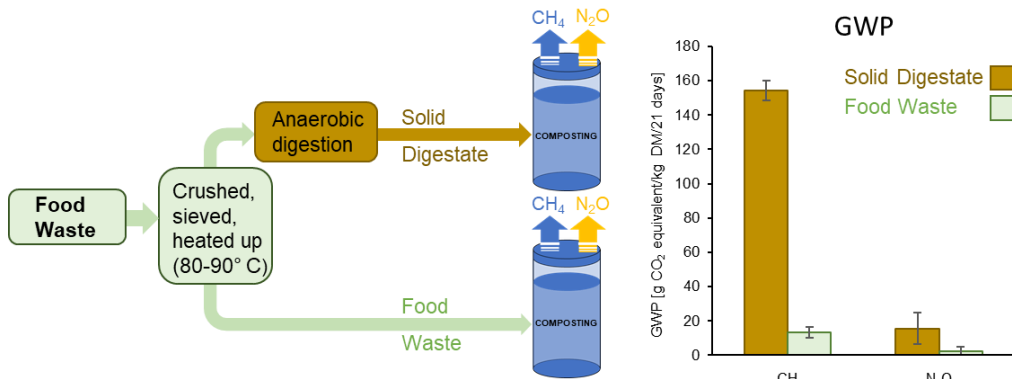
5 ABSTRACT

6 Composting and anaerobic digestion are the most common ways to treat organic
7 residues. Sometimes the organic rest after anaerobic digestion is also composted. In this
8 study we investigated greenhouse gas emissions from composting raw food waste
9 compared to composting solid digestate of food waste. Cumulative methane emissions
10 over 3 weeks were found to be almost 12 times higher from composting digested food
11 waste than from raw food waste suggesting that the microbial community transferred
12 from the anaerobic digestion to the compost process enhanced these emissions.
13 Cumulative nitrous oxide emissions were also higher when composting solid digestate
14 was compared to composting raw food waste, but the global warming potential was
15 mostly driven by the impact of methane emissions. In conclusion, methane production
16 during digestate composting can be high, therefore eliminating methane producing
17 microbes in digestate before composting could be a promising way to reduce
18 greenhouse gas emissions.

19 KEYWORDS

20 Greenhouse gas emission, digestate, food waste, composting, organic waste
21 management

1 GRAPHICAL ABSTRACT



2

3 HIGHLIGHTS

- 4 • Anaerobic digestion before composting increased GWP of N₂O and CH₄
- 5 emissions
- 6 • CH₄ was the major contributor to GWP from composting digested food waste
- 7 • High NH₄ input and an imported microbial community may explain GHG
- 8 emissions

9

1

2 1. Introduction

3 To achieve the well accepted goal of a circular economy, a proper management of
4 organic waste is crucial. Recycling technologies need to be found where greenhouse
5 gas (GHG) emissions to the environment are minimized and nutrients in the organic
6 material returned to the soil. Organic residues contain nutrients, and the decomposition
7 of organic waste also releases energy. The most common ways to treat organic waste
8 are composting and anaerobic digestion (AD). Composting is aerobic degradation, and
9 the energy produced as a microbial by-product during the degradation of organic matter
10 is released as heat. Heat production is caused by microbial activity and therefore
11 depends on moisture, aeration, and C/N ratio. Temperature in the compost is also
12 dependent on ambient temperature and the size and shape of the composting system.
13 Proper aeration is needed for a good composting process, but this also means that
14 ammonia and other gases can be lost during the process. AD produces biogas that
15 contains methane (CH₄) and can be used for energy generation. There is almost no loss
16 of nutrients or gas emission during the process as it happens in a closed container, but
17 there is a risk of losses during later handling of the organic rest, digestate, e.g., via NH₃,
18 N₂O and CH₄ emissions. Digestate, is a good fertilizer with ample plant available
19 nitrogen (Foereid et al., 2021; Odlare et al., 2014; Sogn et al., 2018). Because the water
20 content in digestate is high, above 90 %, it is often separated into a solid fraction and
21 liquid fraction to ease storage and transport. In some cases, the solid digestate (DS) is
22 also composted before application as a soil amendment.

23 Global warming is a world-wide concern. Most organic waste management releases
24 some GHGs (Andersen et al., 2010; Swati and Hait, 2018), so it is important to
25 minimise this as much as possible. Minimising GHG emissions from digestate treatment
26 is important to promote sustainable cities and communities (SDG 11) and make their

1 consumption and production sustainable (SDG 12). It is also a climate action (SDG 13)
2 both because it reduces GHG and because it promotes the use of waste for biogas.

3 Three GHG's are emitted from decomposition processes: carbon dioxide (CO_2), CH_4
4 and nitrous oxide (N_2O). CO_2 returns recently fixed carbon to the atmosphere and is
5 therefore not a net addition. It is produced when microorganisms break down organic
6 matter and therefore CO_2 emissions often serves as an indication of the degradation.

7 Strictly anaerobic methanogenic archaea use carbon (e.g., in form of CO_2 or acetic acid)
8 instead of oxygen as a terminal electron acceptor and produce CH_4 . Therefore, CH_4 is
9 produced in anaerobic processes, but also during composting, as anaerobic zones will
10 usually occur. Some of the produced CH_4 may then be oxidized e.g., by methanotrophic
11 bacteria, before it is emitted to the atmosphere. N_2O is produced during microbial
12 transformations of ammonium (NH_4^+) and nitrate (NO_3^-): NH_4^+ is transformed into
13 nitrite and then NO_3^- (nitrification) with some loss as N_2O , and NO_3^- is then transformed
14 into nitrogen gas (N_2) via denitrification, also with some loss as N_2O . How large the
15 losses as N_2O are in each step, depends on the conditions, but because there are two
16 processes involved requiring different conditions, N_2O emissions can be difficult to
17 predict. Nitrification and denitrification can be performed by various microorganisms
18 such as archaea, bacteria or fungi. Ammonia-oxidizing bacteria and archaea are known
19 to perform the first step of NH_4^+ oxidation while nitrite oxidizing bacteria perform the
20 further oxidation to NO_3^- . Denitrification is mainly known to be facilitated by
21 heterotrophic bacteria but there are also autotrophic denitrifiers have been identified.

22 GHG emissions during composting of organic waste have been assessed. Substantial
23 emissions of both CH_4 and N_2O have been found (Ermolaev et al., 2015;
24 Zhu-Barker et al., 2017). There are only a few studies of GHG emissions during
25 digestate composting (Li et al. (2018) and Zeng et al. (2016)). Dietrich et al. (2020)

1 found that GHG emissions after applications of digestate to soil can be substantial, but
2 applications of composted DS does not induce emissions. That raises the question of
3 how large emissions from composting DS may be. Emissions of N₂O may well be
4 higher from DS because of an increased NH₄⁺ to total nitrogen (N) ratio after AD.
5 Emissions of CH₄ are usually assumed to be lower after AD because most of the CH₄
6 potential has been used up during the AD process (Brémond et al., 2021; Vergote et al.,
7 2020). However, there are also some arguments why CH₄ emissions may be higher from
8 digestate composting than from composting the organic waste directly. Digestate
9 contains a microbial community that is adapted for high CH₄ production (He et al.,
10 2000; Sundberg et al., 2013) and the pH in digestate is elevated and more suitable than
11 in the original residues (Kheiredine et al., 2014). Sometimes there is also some CH₄
12 potential left in the digestate because the digestion process is not run to completion (Li
13 et al., 2020). In addition, fresh organic material, usually added as bulking agents to
14 provide structure (Ahn et al., 2011; Beck-Friis et al., 2000; Bustamante et al., 2013),
15 may provide additional available carbon as CH₄ potential.

16 The objective of this paper was to compare raw food waste composting to digested food
17 waste composting. The composting process and emissions of the GHGs N₂O and CH₄
18 were compared.

19

2. Materials and methods

2.1. Composting feedstocks

Romerike biogas plant (RBA) (60.18728, 11.39981) is a biogas plant treating food waste mainly collected from households in Oslo. The organic waste was grinded, sieved, screw pressed and heated up to 80°-100°C before it was sampled as food waste (FW) in the biogas plant. The biogas plant in Romerike uses a thermal hydrolysis process as pre-treatment and a mesophilic process (38°C) for the AD. DS was sampled after adding a polymer for flocculation and de-watering by a decanter centrifuge. Crushed garden waste (GW) cut down to pieces < 2 cm and stored for half a year was used as structure material. Mature compost (2 months and 3 weeks old) from a mixture of sewage-based digestate and GW was used as inoculum. All the samples and the inoculum were stored at a temperature of 4°C and analyzed by Eurofins using their standard methods. Carbon content was determined on a CHN analyzer (Elementar Vario EL with TCD detector) (Table 1).

Table 1: Characteristics of treatments and structure material: Solid Digestate (DS), Food Waste (FW) and Garden Waste (GW)

Inputs	pH	Conductivity	DM	LOI	Ctot	Ntot	NH ₄ -N	NO ₃ -N
	[]	[mS/m]	[%]	[% DM]	[g Ctot/ 100g DM]	[g Ntot/ 100g DM]	[mg NH ₄ -N/ 100g DM]	[mg NO ₃ -N/ 100g DM]
DS	8.6	150	28.2	73	50.1 ^a	4.6	3.6	< 0.65
FW	4.0	180	10.8	88	45.7 ^a	3.9	< 0.92	18.0
GW	7.4	27	65.2	55	22.6 ^a	0.9	1.2	< 0.29

^a analyzed by NIBIO

2.2. Experimental setup

The same volume of FW or DS (530 ml per container) were mixed with the amount of structure material and water needed to achieve a dry matter content (DM) of +/- 45 % in both mixtures (Table A1 in appendix). The same amount of inoculum per container (27.8 g) was added to the solid digestate mix (DSmix) and the food waste mix

1 (FWmix). Cylindrical Dewar Flasks with a total volume of 2000 ml each and a diameter
2 of 100 mm (see Fig. A1 in appendix), were used. After the feedstocks (DSmix and
3 FWmix) were filled into the containers (3 replicates of each treatment), thermocouples
4 were inserted in combination with a Delta-T Data-logger to measure temperature inside
5 the compost every 10 minutes. HOBO Pendant Temperature/Light 8K data loggers were
6 put on top of the compost to measure the temperature in the headspace. Aeration of the
7 compost was possible through the open top of the Dewar vessel which was covered
8 loosely with aluminium foil to avoid heat and moisture losses. Additionally, the
9 compost was flushed with fresh air through a tube mounted adjacent to the temperature
10 sensor on every gas sampling day. The flasks were kept at ambient temperature
11 ($19^{\circ}\text{C} \pm 1$).

12 2.3. Gas measurements: CO_2 , CH_4 , N_2O

13 Gas measurements were taken over 3 weeks. Gas sampling was performed every day in
14 the beginning and at longer time intervals towards the end of the experiment. Gas
15 concentrations were measured inside the compost and gas emissions were determined as
16 difference between two measurements in the headspace of the temporarily closed
17 container.

18 2.3.1. *Gas concentration sampling inside the compost*

19 A gas sample of approximately 25 ml from inside the compost was pumped into a gas
20 bag using a GA2000 Landfill Gas Analyser (Geotechnical Instruments Ltd. UK). 15 ml
21 of the gas in the bag was then transferred to an evacuated vial using a syringe. For the
22 first 6 sampling days a reading of O_2 concentration was taken with the GA2000 Landfill
23 Gas Analyser.

24 2.3.2. *Headspace sampling*

25 Right after gas concentration sampling inside the compost the aluminium foil was
26 removed. The compost was flushed with ambient air for 2 min through the tube inside

1 the compost using a flow rate of 300 ml/min to ensure O₂ supply and to have equal
2 conditions in all containers before starting the headspace sampling. The tube and the
3 thermocouple were then removed very carefully. After the flushing procedure, the
4 container was closed with an airtight lid and the O₂-measurement taken out through the
5 septum with a syringe (15 ml) and transferred to an evacuated vial. After 10 minutes
6 (and towards the end of the experiment after 20 minutes), the second sample of 15 ml
7 was taken using the same procedure. The gas emission over the period was calculated as
8 the difference between the two. A measurement of height from the compost to the top of
9 the container was taken with a ruler and used to calculate the volume of the headspace.
10 Together with the temperature this was used to calculate gas concentrations in the
11 headspace.

12 2.3.3. GC-MS Analysis

13 The samples in the vials were analyzed by gas chromatography mass spectrometry
14 (GC-MS) to determine concentrations of N₂O, CH₄ and CO₂. The analysis was
15 performed using an Agilent Technologies 7820A GC System gas chromatograph,
16 coupled to a mass detector Agilent Technologies 5875 Series MSD and a Gilson 222
17 XL auto sampler. The sample was injected by a 5 ml sample loop, through a 0.5 m x
18 0.32 mm deactivated precolumn, into a 25 m x 0.32 mm CP-PoraPLOT Q-HT column
19 (Chrompack), kept at 40 °C. Helium was used as carrier gas at 1.0 ml min⁻¹. The
20 mass detector was used in selected ion monitoring (SIM) mode to achieve sufficient
21 sensitivity. A certified mixture of greenhouse gases in helium (AGA) was used as a
22 quality control of the calibration curve (Ekeberg et al. 2004). The concentration in the
23 AGA mixture with the corresponding values obtained in the test in brackets: CH₄: 1.02
24 ± 5.0 % (1.39 ± 20.0 %), CO₂: 991 ± 2.0 % (983 ± 11.0 %) and N₂O: 0.956 ± 5.0 %
25 (1.09 ± 12.0 %).

2.4. Calculations and statistics

2.4.1. Cumulative emissions and global warming potential (GWP)

The emission rates for CH₄-C, CO₂-C and N₂O-N over 3 weeks were calculated for each of the three replicates and then averaged. The trapeze approach was applied to calculate the gas produced between the time steps before cumulating the emissions.

The data set of cumulative emissions from soil after digestate application over 14 days in incubation bottles from Dietrich et al. (2020) was used to compare GWPs of soil application to the results from this study. In that study, food waste and solid digestate for that study was sampled at the same plant in Romerike. The composted solid digestate was collected from ØRAS - Miljøstasjon which received solid digestate from RBA for composting. To compare emissions from soil application with composting process emissions, cumulative emissions were re-calculated to emissions per dry matter content of the feedstock/fertilizer.

The contribution of the GHGs to global warming were calculated using the IPCC 2013 factors (Myhre et al., 2013) for 20 years: 264 for N₂O and 84 for CH₄. The GWPs [g CO₂ eq/kg DM/experimental period] were calculated by

$$GWP = \frac{\text{gas_cum}}{m_F * DM_F} * 0.1 * f_GWP$$

where gas_cum [μg CH₄/experimental period] or [μg N₂O/experimental period] are the cumulative emissions over the experimental period of the specific gas, m_F [g] is the mass of the added feedstock/fertilizer, DM_F its dry matter content in [%] and f_GWP [-] the IPCC 2003 factor or the specific gas for 20 years.

2.4.2. Statistical analysis

Minitab v19.2 statistic software was used. A two-sample t-test was applied to compare emissions at each time point, cumulative emissions and GWP between the FWmix and the DSMix composting. Differences were considered significant when $p < 0.05$.

3. Results and discussion

3.1. Cumulative CH₄ and N₂O emissions

Anaerobic pre-treatment of the feedstock for composting enhanced both cumulative

CH₄-C and N₂O-N emissions, compared to composting raw food waste: Cumulative

CH₄-C emissions after 3 weeks were 12 times higher from the DSmix than from the

FWmix ($p < 0.001$) (Fig 1). Cumulative N₂O-N emissions were almost 7 times higher

for the DSmix than for the FWmix ($p = 0.004$).

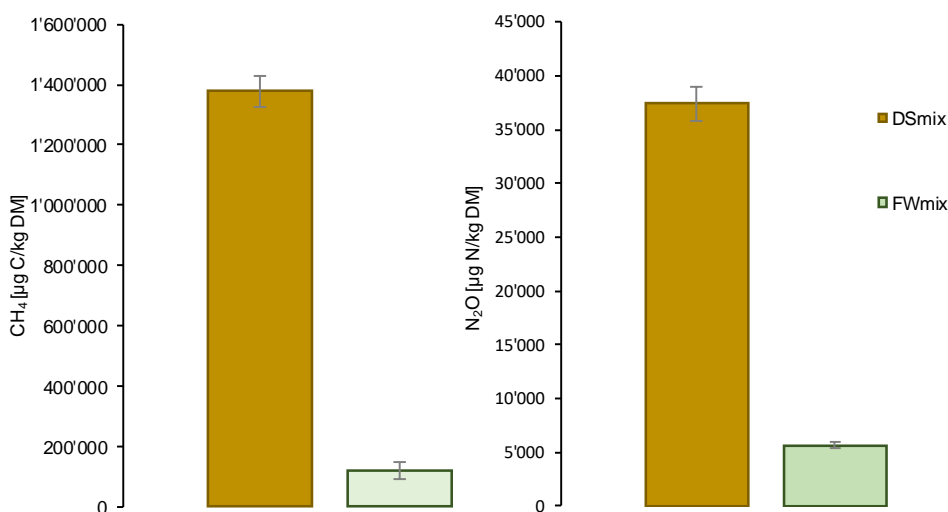


Fig 1: Cumulative total CH₄-C and N₂O-N emission over 3 weeks of composting the FWmix and the DSmix; Error bars indicate standard error (n=3)

3.2. Composting process - Temperature, CO₂, pH, C/N, EC

Temperature is an important indicator for the composting process as it is closely linked

to microbial activity and the decomposition process. The thermophilic (>40°C)

composting process started earlier in the DSmix (after 1 day) than in the FWmix (after

2.5 days) (Fig 2). The highest temperatures were reached between day 4 and 5 in both

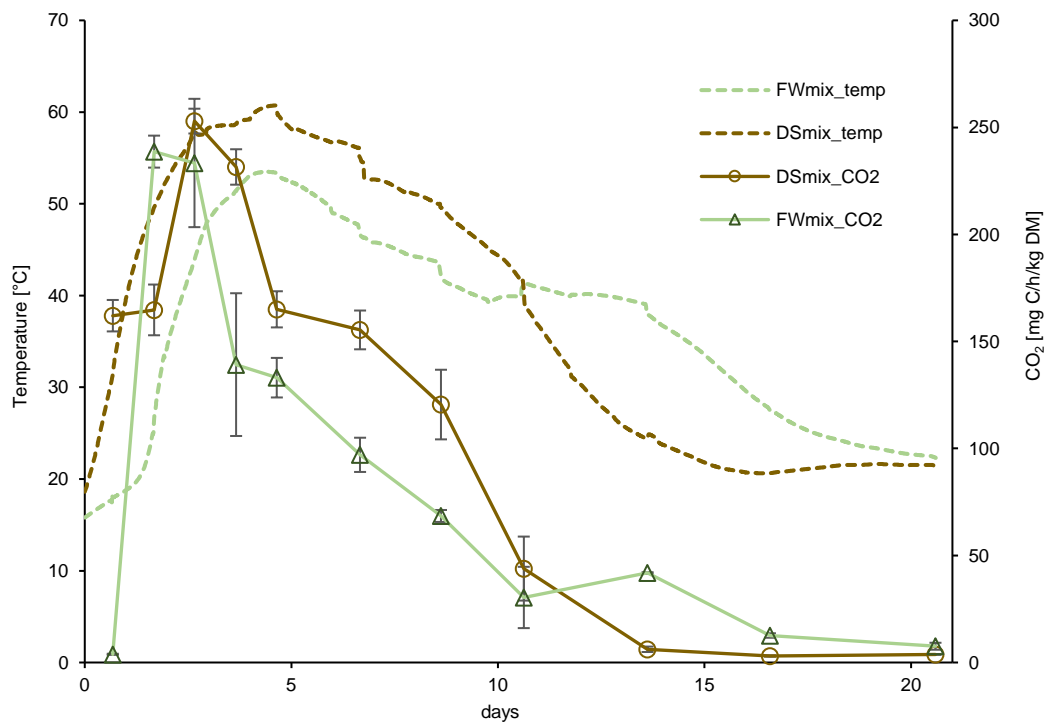
mixtures (53.5°C and 60.7°C for FWmix and DSmix, respectively). The DSmix reached

higher temperatures than the FWmix, but the FWmix stayed thermophilic for a longer

time-period (until day 13 vs. day 10). Both temperatures were near ambient after the 3

weeks indicating the composting process had ended in both mixtures by then.

1 The progression of CO₂ emissions closely followed that of temperature, also reflecting
 2 the development of the composting process. Emissions from the DSmix rose quickly in
 3 the beginning and stayed above those from the FWmix until day 10 (Fig 2). The sharp
 4 immediate increase in temperature and CO₂ emissions from the DSmix suggests that the
 5 AD process was not run to completion and some of the available carbon was still left in
 6 the DS.



7
 8 Fig 2: Temperature measured inside the compost and CO₂ emission rates based on headspace measurements of the
 9 FWmix and the DSmix; Error bars indicate standard error (n=3)

10 pH at the start of the composting process was 4 units lower in the FWmix than in the
 11 DSmix (Table 2). The low pH in the FWmix coincided with lower temperatures than in
 12 the DSmix at the beginning (Fig 2). This confirms similar findings of Sundberg et al.
 13 (2013) who also found a delay in the start-up-phase at low pH. At a low pH (< 6) the
 14 composting process can be inhibited, and this inhibitory effect is then reflected in the
 15 reduction of CO₂ emission (Wong et al., 2009). The high pH (9.1) of the DSmix at the
 16 beginning coincided with an immediate raise in temperature and high CO₂ emissions,
 17 although this pH was higher than what is normally assumed to be optimal for microbial

1 activity. While pH in organic material normally rises during composting, composting
 2 digestate often results in a decrease of pH during the composting process as it starts at
 3 an alkaline pH (Meng et al., 2020; Wang et al., 2017).

4 C/N ratio is known as an indicator for N availability in composting feedstock. The C/N
 5 ratio in the FWmix was higher than in the DSmix before the experiment and lower than
 6 in the DSmix after the experiment (Table 2). Higher C/N ratios than 25 in the starting
 7 material were shown to decrease the decomposition rate while lower values enhance N
 8 losses (Azim, 2018). Meng et al. (2020) claimed that the ideal C/N ratio of composting
 9 feedstock is between 20 and 25 and a lower ratio could slow down the degradation
 10 processes. In our experiment the DSmix achieved higher temperatures although its
 11 initial C/N ratio was lower than in the FWmix and lower than 20. The optimal C/N ratio
 12 in digestate may be different from that in fresh organic material as digestate is already
 13 partly microbially degraded.

14 Table 2: C/N ratios and pH of inputs before composting and mixtures before and after
 15 composting.

material	C/N before composting [-]	C/N after composting [-]	pH before composting [-]	pH after composting [-]
FW	13.3		4.0	
GW	26.3		7.4	
DS	11.5		8.6	
FWmix	21.3	13.5	5.1	8.1
DSmix	17.0	14.3	9.1	8.7

16

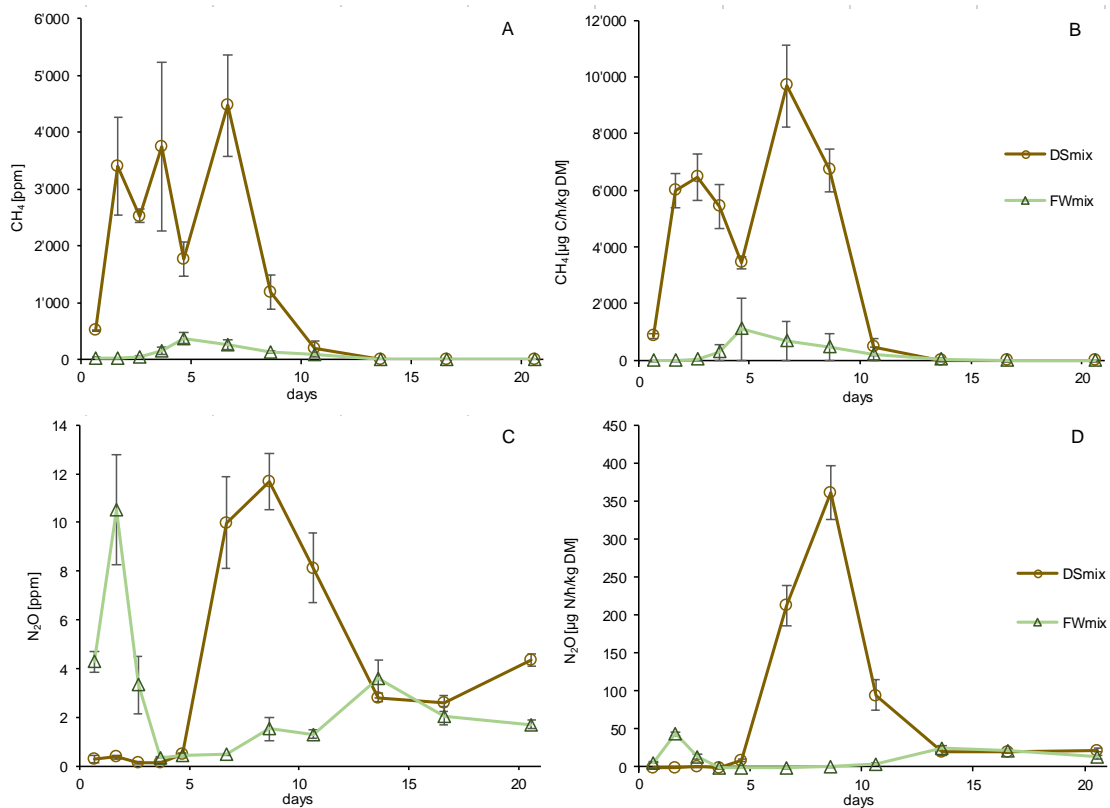
17 CO₂ production is not only used as an index to measure microbial activity but also
 18 maturity of compost (Barrena Gómez et al., 2006). The low emissions from the DSmix
 19 after 11 days and from the FWmix after 14 days indicate that the compost had already
 20 achieved a high degree of stability at the end of the experiment (Fig 2). The low C/N
 21 ratios near to the value of 12 (Fourti 2013) and pH values slightly above neutral (Ferrer

1 et al. 2001) in both mixtures at the end of the experiment (Table 2) also indicate a stable
2 compost in both mixtures.

3 Both initial and both final EC values were below the threshold of 400 mS/m to avoid
4 negative impacts on microorganisms (Zhang and Sun, 2015).

5 3.3. CH₄ production and emission

6 CH₄ emissions from the DSmix were significantly higher than from the FWmix from
7 the first measurement until day 8 (Fig 3, B and D). Emissions from the DSmix were
8 high from the beginning and stayed high until day 8 while emissions from the FWmix
9 were very low over the whole experimental period. This also applied to the
10 concentration measurements inside the compost (Fig 3, A and C) indicating that the
11 difference was caused by differences in CH₄ production and not by uptake.



12

13 Fig 3: CH₄ concentrations measured inside the compost (A) and CH₄ emission rates based on headspace
14 measurements (B) of the DSmix and the FWmix; N₂O concentration inside the compost (C) and N₂O emission rates
15 based on headspace measurements (D) of the DSmix and the FWmix; Error bars indicate standard error (n=3)

16

1 The high CH₄ emissions from the beginning from the DSmix confirm that there was
2 some CH₄ potential left although AD should have used up most of it in DS. Addition of
3 GW and herewith fresh organic material could have provided additional available
4 carbon for CH₄ production.

5 From the DSmix, CH₄ emissions were highest when the temperature was above 38°C,
6 the temperature used during the AD process (Fig 2 and Fig 3). This could indicate that
7 the methanogens present in the DS from the AD process found optimal conditions. After
8 day 10 the temperature dropped quickly far below 38°C again leading to an immediate
9 decrease in CH₄ emissions. In contrast, CH₄ emissions from the FWmix stayed low
10 during the whole experimental period, although the temperature did rise like in the
11 DSmix above 38°C for some days (Fig 2 and Fig 3). One explanation is that the FWmix
12 would have contained far less methanogenic microbes at the start of the experiment.

13 Methanogens generally have extremely low growth rates, for example the doubling time
14 for Methanosaeta lies between 4-9 days (Liu et al., 2011). The population of
15 methanogens would therefore have stayed very small in the FWmix throughout the
16 whole experimental period, explaining the low CH₄ emissions. This interpretation is
17 supported by He et al. (2000) who observed that the addition of mature cattle manure to
18 a food waste compost markedly increased CH₄ emissions after 2 days due to the
19 incorporation of anaerobic microorganisms with the manure.

20 O₂ concentration inside the DSmix was significantly lower than in the FWmix on day 1
21 (see also Fig A2 in appendix). However, the CH₄ production did not start immediately
22 in the DSmix but increased only after almost two days. This increase in CH₄ production
23 after a lag phase was also shown by (Pedizzi et al., 2016) and confirms that CH₄
24 producing microorganisms need some time to start production again after being exposed
25 to unfavourable conditions.

1 3.4. N₂O production and emission
2 N₂O emissions from the DSmix were much higher overall but in the FWmix there was a
3 higher initial peak of emission (Fig 3D). DS had a high NH₄⁺ and low NO₃⁻ content
4 (3.3 and < 0.6 mg/container, resp.), whereas the contrary applied to raw FW (< 0.5 and
5 10.3 mg/container, resp.) (see also Table A2 in appendix). The FWmix contained more
6 than eight times more NO₃-N than the DSmix before composting. The high N₂O
7 concentration in the FWmix in the beginning (Fig 3C) might have been produced via
8 initial denitrification of this high amount of available NO₃⁻-N. Moreover, prior to
9 composting more N₂O producing microbes might have existed in the FW than in DS
10 and they emitted N₂O right away. With the increase in temperature after 2.5 days N₂O
11 production in the FWmix was decreasing again. N₂O producing microbes could have
12 been inhibited or inactivated by unfavourable conditions as it was also reported by He et
13 al. (2000). The microbial community in the DSmix was more adapted to higher
14 temperatures and anaerobic conditions and therefore probably not able to produce N₂O
15 at aerobic conditions and moderate temperatures right away.

16 The N₂O emission peak from the FWmix was less pronounced than the inside peak
17 concentrations during the first 3 days of the experiment (Fig 3C and D). This could
18 indicate a further transformation of N₂O during the diffusion through the compost
19 matrix in the FWmix. The results confirm findings of Andersen et al. (2010) who
20 concluded, that pore space concentrations alone are not enough to estimate GHG
21 emissions from composting to the atmosphere in the case of N₂O. Furthermore, this
22 could indicate that e.g. an additional layer of compost could help reducing N₂O
23 emissions.

24 The NH₄⁺ input per container was at least 30% higher in the DSmix than the FWmix
25 (Table A2) but this did not lead to enhanced N₂O production in the beginning indicating

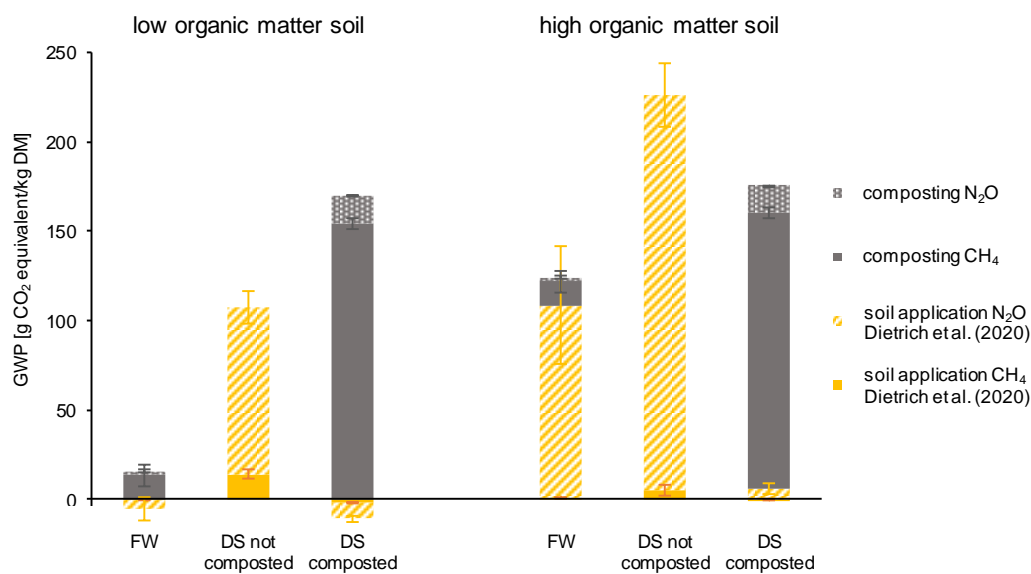
1 that nitrification was not starting right away. Emissions from the DSmix were
2 significantly higher than from the FWmix between day 5 and day 9 and then stayed on a
3 level like that in the FWmix for the rest of the experiment (Fig 3D). Enhanced N₂O
4 emissions were first observed after 5 days when composting the DSmix. Delayed peaks
5 in N₂O emissions after almost one third of the processing time when composting
6 digestate were also found by Zeng et al. (2016). From the DSmix most of the N₂O was
7 emitted during the thermophilic phase and CH₄ and N₂O emissions were both high
8 between day 5 and day 8. The Eh range of N₂O production is known to be higher than
9 the range for CH₄ production. Thus, the emissions of the two gases would generally not
10 be assumed to occur together but concurrent emissions of both gases were also observed
11 by Hao et al. (2004) when composting manure and Jiang et al. (2013) when composting
12 pig feces. This could be explained by high NO₃⁻ content at that stage of the process and
13 subsequent denitrification emitting the N₂O or by microsites providing anaerobic
14 condition for CH₄ production.

15 Ba et al., (2020) and Hao et al., (2001) showed that total nitrogen content of composting
16 substrate is positively related to subsequent N₂O emissions during composting. This
17 positive relation of TN and N₂O emission was also found in our experiment although
18 TN was only 14 % higher in the DSmix than in the FWmix (Table 3).

19 The pH of the FWmix was low at the beginning of the experiment (Table 2) and the
20 initial production of organic acids from the fermentation of carbohydrates and fats may
21 have further decreased pH during the early stages of composting. This could have led to
22 inhibition of nitrification in the FWmix followed by lower N₂O emissions than from the
23 DSmix after 4 days. A high pH is also known to inhibit nitrification (Cayuela et al.,
24 2012) which could have been the reason why N₂O emissions only rose after 5 days in
25 the DSmix.

1 3.5. GWP

2 The GWP of CH₄ and N₂O emissions combined were more than 10 times higher from
3 composting the DSmix than from composting the FWmix ($p = 0.000$) (Fig 4). The
4 comparison of GWPs also revealed that the contribution of CH₄ emissions to total GWP
5 of composting was bigger than the contribution of N₂O emissions. This applied to
6 composting both the DSmix and the FWmix, although the GWP of both gases was
7 much lower for the FWmix.



8

9 Fig 4: GWP of mean cumulated CH₄ and NO₂ emissions from composting process and soil application of FW, DS not
10 composted, and DS composted (low organic matter soil, left; high organic matter soil, right). Error bars indicate
11 standard error (n=3)

12 This high environmental impact of composting DS leads to the question if composting
13 is a suited treatment for digestate. To find an answer, additional information of the
14 whole cycle was included in Fig 4: Dietrich et al. (2020) found high GWP mainly
15 caused by N₂O emissions when DS was applied to the soil. Depending on the soil type
16 the GWP of soil application was found to be even higher than the GWP during the
17 composting process of DS in our experiment (Fig 4). In contrast, the application of
18 composted DS had a very low or negative GWP. These results indicate that the lower
19 GHG emissions during soil application could compensate for the high GWP of

1 composting DS, but it should be considered that the organic matter content of the FW
2 input is reduced during AD by half when leaving the system as digestate.

3 3.6 Anaerobic digestion and greenhouse gas emissions

4
5 Composting is a process carried out by microorganisms and therefore influenced by a
6 number of parameters, including temperature, pH, C/N ratio, oxygen availability and
7 moisture of the feedstock. The AD process changes some of these characteristics of the
8 FW. The DM content was similar in both mixtures, but AD increased the pH into the
9 alkaline range and reduced the C/N ratio. Moreover, AD elevated the NH_4^+ to N ratio
10 and as the digestate was not hygienised after the process microbes could be transferred
11 from AD to the composting process. All these changes of characteristics speeded up the
12 composting process at the beginning and temperatures higher than 55°C ensured a
13 reduction of pathogens. The composting process resulted in a stable compost. We have
14 also shown that GHG emissions from digestate composting can be substantial. This
15 means that strategies for how to minimize these emissions should be sought. Optimizing
16 the composting process could be an effective way to minimize emissions of GHGs. The
17 results suggest focussing on reducing CH_4 emissions as they have the largest
18 contribution to GWP during composting. Finding strategies to inactivate methanogens
19 (e.g., by hygienization) prior to composting or optimizing O_2 availability in the compost
20 could be promising strategies. Other possibilities may include covering compost heaps
21 with finished compost or other materials where CH_4 oxidizers could live and N_2O
22 uptake could take place. The use of biofilters when solid digestate is stored or
23 composted after AD could also be a promising technology to reduce CH_4 emissions
24 substantially.
25 Further research should focus on understanding the mechanisms for the high GHG
26 emission from digestate composting, primarily methane emission, as well as optimizing

1 measures to reduce emission. Determining the microbial community in digestate, how it
2 varies between biogas plants and how it changes during subsequent composting
3 processes would be a priority. Specifically determining the abundance and activity of
4 CH₄ producing and oxidizing microbes would be useful. This may also point to what
5 measures would be most effective in reducing emissions. It may be possible to design
6 hygienization practices that kill the most emitting microbes with the minimum use of
7 energy or to augment compost for covering with the best microbes for both CH₄ and
8 N₂O oxidation.

9

1 4. Conclusion

2 The digestate composting process was faster and reached higher temperatures in the
3 beginning than food waste composting where the elevated temperature lasted longer.
4 Both CH₄ and N₂O emissions during composting were significantly higher from
5 digested food waste than from raw food waste. CH₄ was found to be the major
6 contributor to GWP while N₂O contributed much less to the GWP. The “import” of
7 different microorganisms through feedstock, esp. methanogens in digestate, can enhance
8 CH₄ production during the subsequent composting process. Although composting DS
9 has a high environmental impact the low GWP after application of composted digestate
10 to soil could compensate.

11

1 **Acknowledgement**

2 This work was funded by the Research Council of Norway through the RICEDIG
3 project [grant number 283570]. The authors thank Romerike Biogas Plant for providing
4 composting feedstocks and Tormod Briseid for useful comments on the manuscript.

5

1 References

- 2
- 3 1. Ahn, H.K., Mulbry, W., White, J.W., Kondrad, S.L., 2011. Pile mixing increases
4 greenhouse gas emissions during composting of dairy manure. *Bioresour.*
5 *Technol.* 102, 2904–2909. <https://doi.org/10.1016/j.biortech.2010.10.142>
- 6 2. Andersen, J.K., Boldrin, A., Samuelsson, J., Christensen, T.H., Scheutz, C.,
7 2010. Quantification of Greenhouse Gas Emissions from Windrow Composting
8 of Garden Waste. *J. Environ. Qual.* 39, 713–724.
9 <https://doi.org/10.2134/jeq2009.0329>
- 10 3. Azim, K., Soudi, B., Boukhari, S., Perissol, C., Roussos, S., Thami Alami, I.,
11 2018. Composting parameters and compost quality: a literature review. *Org.*
12 *Agric.* 8, 141–158. <https://doi.org/10.1007/s13165-017-0180-z>
- 13 4. Ba, S., Qu, Q., Zhang, K., Groot, J.C.J., 2020. Meta-analysis of greenhouse gas
14 and ammonia emissions from dairy manure composting. *Biosyst. Eng.* 193,
15 126-137. <https://doi.org/10.1016/j.biosystemseng.2020.02.015>
- 16 5. Barrena Gómez, R., Vázquez Lima, F., Sánchez Ferrer, A., 2006. The use of
17 respiration indices in the composting process: A review. *Waste Manag. Res.* 24,
18 37–47. <https://doi.org/10.1177/0734242X06062385>
- 19 6. Beck-Friis, B., Pell, M., Sonesson, U., Jönsson, H., Kirchmann, H., 2000.
20 Formation and emission of N₂O and CH₄ from compost heaps of organic
21 household waste. *Environ. Monit. Assess.* 62, 317–331.
22 <https://doi.org/10.1023/A:1006245227491>
- 23 7. Brémond, U., Bertrandias, A., de Buyer, R., Latrille, E., Jimenez, J., Escudié, R.,
24 Steyer, J.P., Bernet, N., Carrere, H., 2021. Recirculation of solid digestate to
25 enhance energy efficiency of biogas plants: Strategies, conditions and impacts.
26 *Energy Convers. Manag.* 231, 113759.

- 1 <https://doi.org/10.1016/j.enconman.2020.113759>
- 2 8. Bustamante, M.A., Restrepo, A.P., Albuquerque, J.A., Pérez-Murcia, M.D.,
3 Paredes, C., Moral, R., Bernal, M.P., 2013. Recycling of anaerobic digestates by
4 composting: Effect of the bulking agent used. *J. Clean. Prod.* 47, 61–69.
5 <https://doi.org/10.1016/j.jclepro.2012.07.018>
- 6 9. Cayuela, M.L., Sánchez-Monedero, M.A., Roig, A., Sinicco, T., Mondini, C.,
7 2012. Biochemical changes and GHG emissions during composting of
8 lignocellulosic residues with different N-rich by-products. *Chemosphere* 88,
9 196–203. <https://doi.org/10.1016/j.chemosphere.2012.03.001>
- 10 10. Dietrich, M., Fongen, M., Foreid, B., 2020. Greenhouse Gas Emissions from
11 Digestate in Soil. *Int. J. Recycl. Org. Waste Agric.* 9, 1–19.
12 <https://doi.org/10.30486/IJROWA.2020.1885341.1005>
- 13 11. Ekeberg, D., Ogner, G., Fongen, M., Joner, E.J., Wickstrøm, T., 2004.
14 Determination of CH₄, CO₂ and N₂O in air samples and soil atmosphere by gas
15 chromatography mass spectrometry, GC-MS. *J. Environ. Monit.* 6, 621–623.
16 <https://doi.org/10.1039/b401315h>
- 17 12. Ermolaev, E., Jarvis, Å., Sundberg, C., Smårs, S., Pell, M., Jönsson, H., 2015.
18 Nitrous oxide and methane emissions from food waste composting at different
19 temperatures. *Waste Manag.* 46, 113–119.
20 <https://doi.org/10.1016/j.wasman.2015.08.021>
- 21 13. Ferrer, J., Páez, G., Mármol, Z., Ramones, E., Chandler, C., Marín, M., Ferrer,
22 A., 2001. Agronomic use of biotechnologically processed grape wastes.
23 *Bioresour. Technol.* 76, 39–44. [https://doi.org/10.1016/S0960-8524\(00\)00076-6](https://doi.org/10.1016/S0960-8524(00)00076-6)
- 24 14. Foereid, B., Szocs, J., Patinvoh, R.J., Horváth, I.S., 2021. Effect of anaerobic
25 digestion of manure before application to soil – benefits for nitrogen utilisation?
26 *Int. J. Recycl. Org. Waste Agric.* 10, 89–99.

- 1 <https://doi.org/10.30486/IJROWA.2020.1897538.1055>
- 2 15. Fourti, O., 2013. The maturity tests during the composting of municipal solid
3 wastes. *Resour. Conserv. Recycl.* 72, 43–49.
4 <https://doi.org/10.1016/j.resconrec.2012.12.001>
- 5 16. Hao, X., Chang, C., Larney, F.J., Travis, G.R., 2001. Greenhouse Gas Emissions
6 during Cattle Feedlot Manure Composting. *J. Environ. Qual.* 31, 376–386.
7 <https://doi.org/10.2134/jeq2002.7000>
- 8 17. Hao, X., Chang, C., Larney, F.J., 2004. Carbon, Nitrogen Balances and
9 Greenhouse Gas Emission during Cattle Feedlot Manure Composting. *J.*
10 *Environ. Qual.* 33, 37. <https://doi.org/10.2134/jeq2004.0037>
- 11 18. He, Y., Inamori, Y., Mizuochi, M., Kong, H., Iwami, N., Sun, T., 2000.
12 Measurements of N₂O and CH₄ from the aerated composting of food waste. *Sci.*
13 *Total Environ.* 254, 65–74. [https://doi.org/10.1016/S0048-9697\(00\)00439-3](https://doi.org/10.1016/S0048-9697(00)00439-3)
- 14 19. Jiang, T., Schuchardt, F., Li, G.X., Guo, R., Luo, Y.M., 2013. Gaseous emission
15 during the composting of pig feces from Chinese Ganqinfen system.
16 *Chemosphere* 90, 1545–1551.
17 <https://doi.org/10.1016/j.chemosphere.2012.08.056>
- 18 20. Kheiredine, B., Derbal, K., Bencheikh-Lehocine, M., 2014. Effect of starting pH
19 on the produced methane from dairy wastewater in thermophilic phase. *Chem.*
20 *Eng. Trans.* 38, 511–516. <https://doi.org/10.3303/CET1438086>
- 21 21. Li, Y., Luo, W., Lu, J., Zhang, X., Li, S., Wu, Y., Li, G., 2018. Effects of
22 digestion time in anaerobic digestion on subsequent digestate composting.
23 *Bioresour. Technol.* 267, 117–125.
24 <https://doi.org/10.1016/j.biortech.2018.04.098>
- 25 22. Li, Y., Han, Y., Zhang, Y., Fang, Y., Li, S., Li, G., Luo, W., 2020. Factors
26 affecting gaseous emissions, maturity, and energy efficiency in composting of

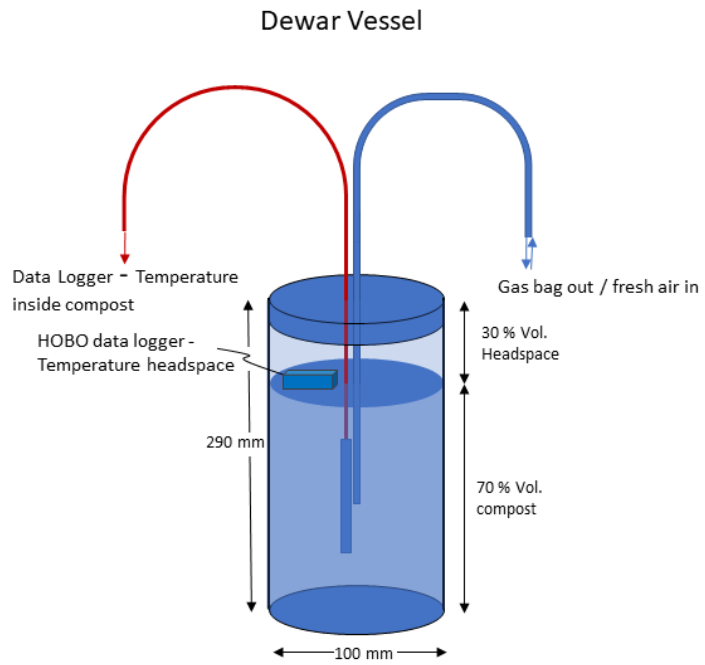
- 1 livestock manure digestate. *Sci. Total Environ.* 731, 139157.
2 <https://doi.org/10.1016/j.scitotenv.2020.139157>
- 3 23. Liu, X., Yan, Z., Yue, Z.-B., 2011. 3.10 - Biogas, in: Moo-Young, M. (Ed.),
4 *Comprehensive Biotechnology (Second Edition)*. Academic Press, pp. 99–114.
5 <https://doi.org/10.1016/B978-0-08-088504-9.00165-3>
- 6 24. Meng, X., Yan, J., Zuo, B., Wang, Y., Yuan, X., Cui, Z., 2020. Full-scale of
7 composting process of biogas residues from corn stover anaerobic digestion:
8 Physical-chemical, biology parameters and maturity indexes during whole
9 process. *Bioresour. Technol.* 302, 122742.
10 <https://doi.org/10.1016/j.biortech.2020.122742>
- 11 25. Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestedt, J., Huang, J.,
12 Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A.,
13 Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and natural
14 radiative forcing. In: *Climate Change 2013: The Physical Science Basis*.
15 Contribution of Working Group I to the Fifth Assessment Report of the
16 Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K.
17 Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and
18 P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom
19 and New York, NY, USA. 659–740. <https://doi.org/10.3390/jmse6040146>
- 20 26. Odlare, M., Pell, M., Arthurson, J. V., Abubaker, J., Nehrenheim, E., 2014.
21 Combined mineral N and organic waste fertilization-effects on crop growth and
22 soil properties. *J. Agric. Sci.* 152, 134–145.
23 <https://doi.org/10.1017/S0021859612001050>
- 24 27. Pedizzi, C., Regueiro, L., Rodriguez-Verde, I., Lema, J.M., Carballa, M., 2016.
25 Effect of oxygen on the microbial activities of thermophilic anaerobic biomass.
26 *Bioresour. Technol.* 211, 765–768.

- 1 <https://doi.org/10.1016/j.biortech.2016.03.085>
- 2 28. Sogn, T.A., Dragicevic, I., Linjordet, R., Krogstad, T., Eijsink, V.G.H., Eich-
3 Gretorex, S., 2018. Recycling of biogas digestates in plant production: NPK
4 fertilizer value and risk of leaching. *Int. J. Recycl. Org. Waste Agric.* 7, 49–58.
5 <https://doi.org/10.1007/s40093-017-0188-0>
- 6 29. Sundberg, Carina, Al-Soud, W.A., Larsson, M., Alm, E., Yekta, S.S., Svensson,
7 B.H., Sørensen, S.J., Karlsson, A., 2013. 454 Pyrosequencing Analyses of
8 Bacterial and Archaeal Richness in 21 Full-Scale Biogas Digesters. *FEMS*
9 *Microbiol. Ecol.* 85, 612–626. <https://doi.org/10.1111/1574-6941.12148>
- 10 30. Sundberg, Cecilia, Yu, D., Franke-Whittle, I., Kauppi, S., Smårs, S., Insam, H.,
11 Romantschuk, M., Jönsson, H., 2013. Effects of pH and microbial composition
12 on odour in food waste composting. *Waste Manag.* 33, 204–211.
13 <https://doi.org/10.1016/j.wasman.2012.09.017>
- 14 31. Swati, A., Hait, S., 2018. Greenhouse Gas Emission During Composting and
15 Vermicomposting of Organic Wastes – A Review. *Clean - Soil, Air, Water*
16 46 (6), p.1700042. <https://doi.org/10.1002/clen.201700042>
- 17 32. Vergote, T.L.I., Bodé, S., De Dobbelaere, A.E.J., Buysse, J., Meers, E., Volcke,
18 E.I.P., 2020. Monitoring methane and nitrous oxide emissions from digestate
19 storage following manure mono-digestion. *Biosyst. Eng.* 196, 159–171.
20 <https://doi.org/10.1016/j.biosystemseng.2020.05.011>
- 21 33. Wang, T.T., Wang, S.P., Zhong, X.Z., Sun, Z.Y., Huang, Y.L., Tan, L., Tang,
22 Y.Q., Kida, K., 2017. Converting digested residue eluted from dry anaerobic
23 digestion of distilled grain waste into value-added fertilizer by aerobic
24 composting. *J. Clean. Prod.* 166, 530–536.
25 <https://doi.org/10.1016/j.jclepro.2017.08.075>
- 26 34. Wong, J.W.C., Fung, S.O., Selvam, A., 2009. Coal fly ash and lime addition

- 1 enhances the rate and efficiency of decomposition of food waste during
2 composting. *Bioresour. Technol.* 100, 3324–3331.
3 <https://doi.org/10.1016/j.biortech.2009.01.063>
- 4 35. Zeng, Y., De Guardia, A., Dabert, P., 2016. Improving composting as a post-
5 treatment of anaerobic digestate. *Bioresour. Technol.* 201, 293–303.
6 <https://doi.org/10.1016/j.biortech.2015.11.013>
- 7 36. Zhang, L., Sun, X., 2015. Effects of earthworm casts and zeolite on the two-
8 stage composting of green waste. *Waste Manag.* 39, 119–129.
9 <https://doi.org/10.1016/j.wasman.2015.02.037>
- 10 37. Zhu-Barker, X., Bailey, S.K., Paw U, K.T., Burger, M., Horwath, W.R., 2017.
11 Greenhouse gas emissions from green waste composting windrow. *Waste*
12 *Manag.* 59, 70–79. <https://doi.org/10.1016/j.wasman.2016.10.004>
- 13

1 Appendix

2



3

4 Fig A1: Experimental setup for gas measurement with Dewar Vessel

5

6

1 Table A1: DM and Loss on Ignition (LOI) of solid
 2 Digestate Mix (DSmix) and Food Waste Mix
 3 (FWmix). Values are means (SD). n=3

MIX	DM	LOI
	[%]	[%]
DSmix	44.7 (0.7)	44.3 (1.7)
FWmix	46.1 (0.4)	49.2 (1.8)

4

5

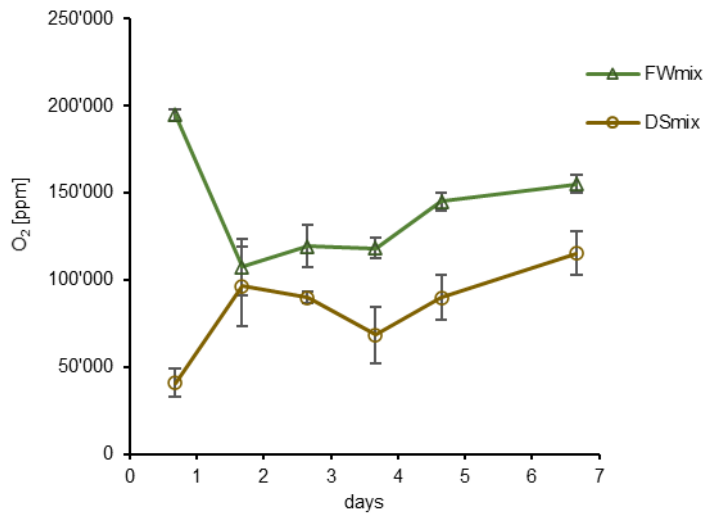
6 Table A2: TC, TN, NH₄-N, NO₃-N and mineral nitrogen (NO₃-N+NH₄-N) inputs per container

	DM added	TC	TN	NH₄-N	NO₃-N	NO₃-N and NH₄-N
	[g]	[g]	[g]	[mg]	[mg]	[mg]
FW	57.2	26.2	2.2	< 0.5	10.3	10.9
GW	399.1	90	3.6	4.8	< 1.1	4.8
FWmix	456.3	116.2	5.8	< 5.3	< 11.5	15.7
DS	92.6	46.4	4.3	3.3	< 0.6	3.3
GW	279.6	63.1	2.5	3.4	< 0.8	3.4
DSmix	372.2	109.5	6.7	6.7	< 1.4	6.7

7

8

1



2

3 Fig A2: Mean O₂ concentration inside the compost during the first week
4 of the DSmix and the FWmix; Error bars indicate standard error (n=3)

5