

GASEOUS EMISSIONS FROM THE BIOLOGICAL VIA NITRITE NITROGEN REMOVAL PROCESS TREATING ANAEROBICALLY CO-DIGESTED EFFLUENTS

N. Frison¹, A. Chiumenti², E. Katsou³, S. Malamis³, D. Bolzonella³, F. Fatone³

¹*Department of Environmental Sciences, Informatics and Statistics, University Ca' Foscari of Venice, Dorsoduro 2137 - 30121 Venice, Italy*

²*DISA Department, University of Udine, Via delle Scienze 208, 33100, Udine, Italy*

³*Department of Biotechnology, University of Verona, Strada Le Grazie 15, 37134, Verona, Italy*

Contact details: e-mail: malamis.simos@gmail.com; Tel/Fax: +39 045 8027965

Abstract

In this work the gaseous emissions of nitrous oxide (N₂O), carbon dioxide (CO₂) methane (CH₄), ammonia (NH₃) and methyl mercaptan (CH₃SH) produced from the nitrification/denitrification biological nutrient removal (BNR) process were determined in a sequencing batch reactor (SBR). The SBR treated the anaerobic supernatant produced from the anaerobic co-digestion of the organic fraction of municipal solid waste (OFMSW) and waste activated sludge (WAS). Two experimental periods were carried out: period 1 in which a volumetric nitrogen loading rate (vNLR) of 0.81 kgN/m³d was applied, using acetic acid as carbon source and period 2 with vNLR=1.09 kgN/m³d using fermentation liquid produced from biowaste as carbon source. N₂O emissions were 0.24% and 1.38% of the influent nitrogen load in periods 1 and 2 respectively. The higher dissolved oxygen (DO= 1.5 mg/L) concentration as well as the lower accumulation of nitrite resulted in significantly lower N₂O emissions in period 1 compared to 2. NH₃ emissions were very low (<1% of influent nitrogen load), while some methane (5.3% of influent load as COD) was emitted during period 2, as the lower DO (0.95 mg/L) concentration probably created micro-anaerobic conditions within the sludge flocs; this was also favoured by the use of fermentation liquid as a carbon source.

Keywords

Gaseous emissions; nitrous oxide emissions, nitrification/denitrification; sequencing batch reactor

Introduction

During the operation of a wastewater treatment plant (WWTP) nitrous oxide (N₂O), carbon dioxide (CO₂), methane (CH₄), nitric oxide (NO) as well as other gases can be emitted. N₂O is of particular environmental concern since it has a global warming potential that is 300 times higher than that of CO₂. The global production of N₂O emissions from wastewater treatment was estimated to be 0.22 TgNO₂-N/yr in 1990, which corresponded to 3.2% of the total estimated anthropogenic N₂O emissions in the world [1-3]. The contribution of biological nutrient removal (BNR) to N₂O emissions can be up to 10.2% [4]. The N₂O emissions from wastewater management are estimated to contribute 26% to the total greenhouse gas emissions of the water chain. The latter includes water supply and treatment, wastewater collection and treatment and final discharge [3, 5]. The IPCC guidelines (2006) decreased the standard N₂O emission factor from 1% to 0.5% of the nitrogen load of the WWTP influent. In countries having advanced

WWTPs, a low factor is applied by IPCC (2006) for direct emissions from WWTPs; this is 3.2 gN/person/year and corresponds to ~0.035% nitrous oxide emissions of the influent nitrogen load of a WWTP [3]. In terms of CO₂ equivalents nitrous oxide contributes by 7.9% to the total anthropogenic greenhouse gas (GHG) emissions. Methane has global warming potential that is 23 times higher than that of CO₂ and contributes by 14% to the total GHG emissions. Methane is mainly emitted from livestock farming activities (166 million tonnes CO₂ equivalent in the EU per year), from landfills, composting units and WWTPs (95 million tonnes CO₂ equivalent in the EU per year) [6, 7].

BNR from wastewater is an effective approach for the prevention of eutrophication of water recipients. However, many practical design and operating decisions in WWTPs (including the BNR processes) have considerable impact on the overall environmental performance, in particular concerning the GHG emissions [8]. The vast majority of nitrous oxide emissions in WWTPs occur in the biological treatment processes. At the level of a BNR treatment plant, the N₂O impact can reach up to 83% of the operational CO₂ footprint [9]. N₂O emissions during BNR occur during the biological processes of nitrification and denitrification (Figure 1). During the nitrification process, N₂O can be formed via two routes: the first pathway is as a by-product of the incomplete oxidation of hydroxylamine (NH₂OH) to nitrite. Hydroxylamine is formed from the oxidation of ammonium from ammonium oxidizing bacteria (AOB) using the enzyme ammonia monooxygenase (AMO) to catalyze the reaction. NH₂OH is then oxidized to nitrite using hydroxylamine oxidoreductase (HAO), a biochemical reaction which produces nitrous oxide. The second pathway of N₂O formation is attributed to the process known as nitrifier denitrification. In this biochemical process nitrite is used as electron acceptor instead of oxygen; this can occur under limiting dissolved oxygen (DO) conditions during nitrification. In this biochemical pathway, nitrite reductase (NIR) and nitric oxide reductase (NOR) catalyze the reduction of NO₂⁻ to NO and N₂O by AOB. The third pathway occurs during the anoxic operation of the reactor. In this process, NO and N₂O are produced as process intermediates of NO₃⁻ reduction to N₂. This is also the only stage in which N₂O is also consumed as it is reduced to N₂ [10]. N₂O can be produced in the anoxic phase and subsequently be stripped to the gas phase in an aerated compartment. As N₂O has a relatively high solubility in water, stripping is therefore not very fast [3].

Desloover et al. [4] performed an overview of the quantified N₂O emissions from full scale BNR plants that apply the conventional nitrification/denitrification and advanced nitrification/anammox and nitrification/denitrification processes; they concluded that nitrification is the bioprocess that mainly contributes to N₂O emissions. Full-scale measurements also point to nitrite as a factor in N₂O production [11]. Taking into account that the much higher greenhouse gas impact of N₂O compared to CO₂, it is necessary to determine whether nitrogen removal bioprocesses based on transient nitrite accumulation are systematically greater contributors of N₂O than full nitrification processes [2, 12]. N₂O emission rates can vary considerably due to the differences in the wastewater composition, the applied treatment process, the operating parameters and the environmental conditions. Figure 2 summarizes the most important parameters that affect the N₂O emissions: these include the DO concentration which should not be low during nitrification and the nitrite levels in the mixed liquor which should not accumulate to high levels. In this sense the use of SBR technology, particularly when combined

with the treatment of highly nitrogenous effluents can enhance the nitrous oxide emissions due to the accumulation of nitrite in the reactor.

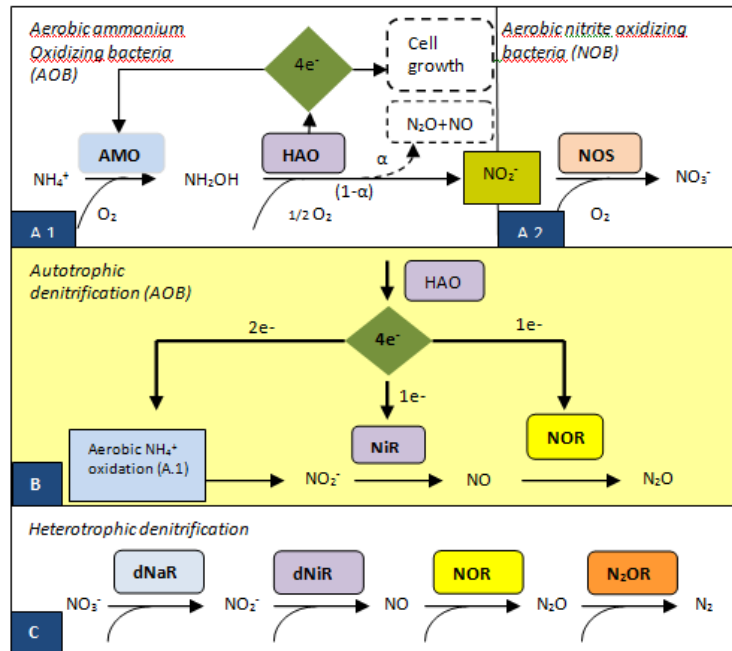


Figure 1: Simplified representation of the biochemical processes which are responsible for nitrous oxide production during nitrification and denitrification: A1 aerobic ammonium oxidation by AOB, A2 aerobic nitrite oxidation by nitrite oxidizing bacteria (NOB), B nitrifier denitrification by AOB and C heterotrophic denitrification [10].

The novel technologies that are introduced should not only remove/recover nutrients at a low cost, but should also be environmentally friendly. Nitritation/denitritation is increasingly being applied, particularly for the treatment of nitrogenous effluents, as it has lower energy requirements for aeration and organic carbon source requirements for denitrification than conventional nitrification/denitrification. However, such processes can result in significant N₂O emissions [13]. The implementation of strategies to mitigate N₂O emissions in the via nitrite processes can increase their sustainability. For example, the use of sludge fermentation liquid as carbon source in the via nitrite processes can mitigate N₂O and NO emissions [14]. Recording and monitoring of the GHG compounds is required to abate their emissions through the implementation of the best available mitigation strategies.

The aim of this work was to determine the gaseous emissions (and particularly nitrous oxide) during the real time operation of a pilot scale nitritation/denitritation process implemented using a sequencing batch reactor (SBR). The SBR treated the anaerobic supernatant produced from the co-digestion of the organic fraction of municipal solid waste (OFMSW) and waste activated sludge (WAS).

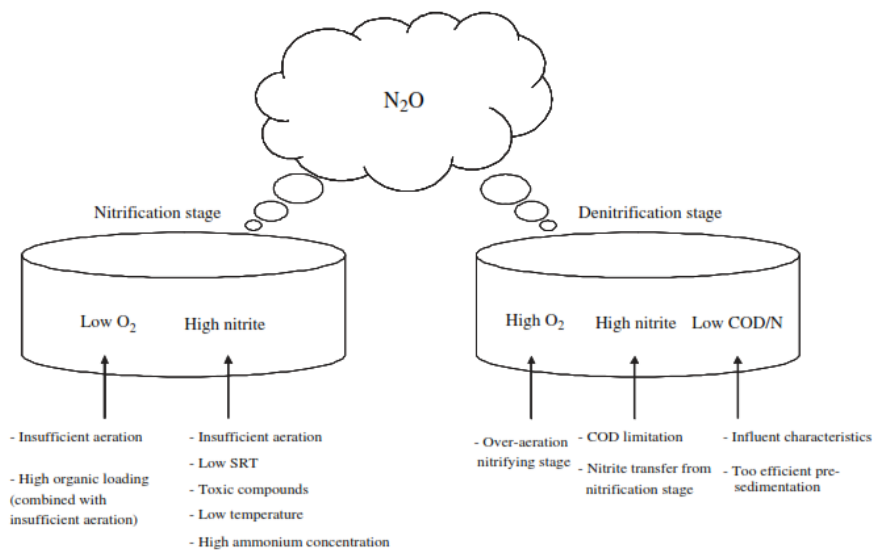


Figure 2: Important parameters leading to N₂O emissions [3]

Material and methods

SBR operation and experimental periods

The pilot scale SBR was installed at the integrated urban municipal waste and wastewater treatment plant (WWTP) of Treviso municipality (Veneto Region), designed to serve a population equivalent of 70,000. The SBR treated up to 6 m³/d of anaerobic supernatant which was produced from the co-digestion of WAS and the OFMSW. The anaerobic digestion took place in a mesophilic (37°C) anaerobic digester (reaction volume 2000 m³) and was fed with 120 m³/d ± 30% of thickened WAS and with 5.1 tonnes/d ± 95% of source separated OFMSW in order to obtain an organic loading rate (OLR) of 1.7 kgVS/m³·d. The hydraulic retention time (HRT) was 20 d. The anaerobic digestate was dewatered from 4-6% total solids (TS) to 20-25% TS. The liquid stream that was produced (i.e. anaerobic supernatant) was sent to a storage tank and from there it was fed to the pilot scale SBR. The reaction volume of the SBR was 2.8 m³ [15]. As the anaerobic supernatant was characterized by a very low COD/N ratio an external organic carbon source was applied in order to supply sufficient carbon for denitrification. The external organic carbon was always added during the first 5 minutes of the anoxic phase.

Agitation was achieved using a Rushton turbine, while three blowers were used for the reactor's aeration. The on/off operation of the blowers was automatically controlled on the basis of the DO and/or pH real-time measurements, properly processed by the programmable logic controller (PLC - Telemecanique, Schneider Electric, Germany). On line submerged probes were used to monitor the SBR operation. These consisted of the DO (LDO Hach Lange), the oxidation reduction potential (ORP, Chemitec and Hach Lange), pH (Hach Lange), conductivity (Hach Lange), NO_x-N (Nitratax plus Hach Lange coupled with the sample filtration system Filtrax - Hach-Lange), NH₄-N (NH₄Dsc, Hach-Lange), mixed liquor suspended solids (MLSS, Solitax, Hach-Lange) and temperature. The heating system was set on when the mixed liquor temperature dropped below 15°C. The on line signals were processed by the PLC where the control algorithms were running. During the start up of the process, the via nitrite nitrogen removal

process was accomplished within 20 days. Complete inhibition of the NOB was obtained with nitrite accumulation being 100% [15].

Then two experimental periods were carried out: in period 1 a volumetric nitrogen loading rate (vNLR) of 0.81 kgN/m³d was applied and acetic acid was used as carbon source at a ratio of 2.1 kgCOD/kg(NO₂-N)_{denitrified}. The sequence of the SBR cycle was: filling, aerobic reaction, anoxic reaction, settling, decanting, idle time. In period 2, the vNLR increased to 1.08 kgN/m³d; the external organic source that was fed consisted of fermentation liquid (FL) produced from source separated organic fraction municipal solid waste (OFMSW) at a ratio of 3.1 kgCOD/kg(NO₂-N)_{denitrified}. Another difference was that the aerobic reaction took place after the anoxic reaction in the SBR cycle.

Analytical methods

TS, total volatile solids (TVS), total and volatile suspended solids (TSS, VSS), MLSS, mixed liquor suspended solids (MLVSS), pH, conductivity, alkalinity, chemical oxygen demand (COD), soluble COD (sCOD), total Kjehldahl nitrogen (TKN), nitrate (NO₃-N), nitrite (NO₂-N), phosphorus (TP) and phosphate (PO₄-P) were determined by Standards Methods [16] and ion chromatography for the anions (Dionex IC-90 with AG14 and AS14 columns)

Determination of gaseous emissions

The determination of the gaseous emissions from the surface of the SBR tank was performed on line by the static chamber method and the Bruel and Kjaer photo-acoustic analyzer (Type 1302, Copenhagen, Denmark) Specifically, a 0.2 m high floating container (chamber) was placed on the surface of the tank and the time variation of the gas concentration inside the chamber was measured on line, in order to determine its emission rate (Figure 3). The gas concentration inside the chamber increased with time as a result of the gaseous emissions from the surface of the reactor [17, 18]. For each gas, a concentration vs. time plot was prepared and an interval of linear increase of the concentration was determined (Figure 4); the emission rate was calculated as follows [19]:

$$E_R = \frac{C_i - C_0}{t_i - t_0} \frac{V_{ch}}{A_{ch}}$$

Where:

E_R (mg/m²h) is the emission rate of the gas

t_i (h) and t_0 (h) represent the time edges of the linear portion of the concentration plot

C_i (mg/m³) and C_0 (mg/m³) represent the gas concentration at times t_i and t_0 respectively

V_{ch} is the volume of the chamber, 0.01116 m³ and

A_{ch} is the area of the emitting surface covered by the chamber, 0.07789 m².

This way E_R was determined for NH₃, N₂O, CO₂, CH₄ and methyl mercaptan (CH₃SH) at various times of the SBR process, during aeration reaction, anoxic reaction and sedimentation. Therefore, a plot of E_R versus time was obtained for each gas for an SBR cycle. To determine the total amount of gas emitted during one cycle, the area below this graph was determined using the following formula:

$$G_M = \sum (E_R A_{SBR} \Delta t)$$

Where:

G_M (mg/cycle) is the amount of the emitted gas per cycle
 A_{SBR} (m²) is the surface area of the pilot SBR, $1.5 \cdot 1.5 = 2.25$ m²
 Δt (h) is the time interval during which the gas emissions were recorded



Figure 3: (a) photo of the configuration of the SBR with the floating container for the on line gas measurement, (b) instrument for the on line measurement of gas emissions and (c) cross section and side view of the floating container

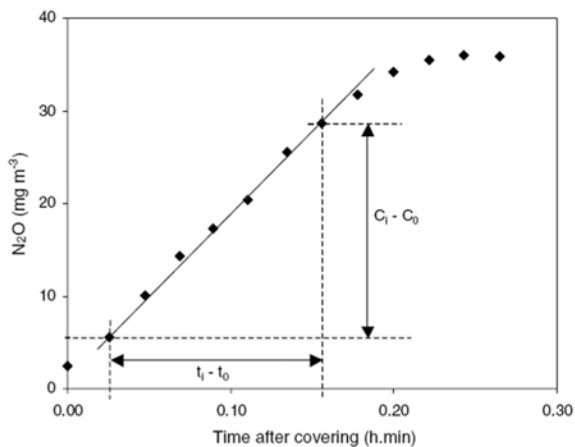


Figure 4: Increase of nitrous oxide concentration with time once the floating container is placed on the surface of the bioreactor (corresponding to one measurement in the SBR cycle)

Results and discussion

Operating periods

The operating characteristics of the two experimental periods are summarized in Table 1 and the physicochemical characteristics of the anaerobic supernatant and the OFMSW FL are shown in Table 2. The vNLR that was applied in period 1 was close to its nitrogen removal capacity. The vNLR applied in period 2 was >30% higher than the system's nitrogen removal capacity.

Table 1: Operating conditions during the experimental period (mean values \pm standard deviation)

Parameters	Units	Period 1	Period 2
vNLR	kgN/m ³ ·d	0.81 \pm 0.14	1.08 \pm 0.20
sNLR	kgN/kgMLVSS·d	0.29 \pm 0.06	0.54 \pm 0.10
HRT	d	0.75 \pm 0.01	0.52 \pm 0.15
OLR	kgCOD/m ³ ·d	1.68	2.13
F/M	kgCOD/kgMLVSS·d	0.60	1.08
MLSS	g/L	3.62 \pm 1.26	2.65 \pm 0.47
MLVSS	g/L	2.78 \pm 1.41	1.98 \pm 0.49
FA	mgNH ₃ -N/L	3.22 \pm 1.84	7.14 \pm 6.20
FNA	mgHNO ₂ -N/L	<0.02	<0.02
DO	mgO ₂ /L	1.48 \pm 0.07	0.95 \pm 0.12

HRT: hydraulic retention time, OLR: organic loading rate, F/M: food to microorganism ratio, FA: free ammonia, FNA: free nitrous acid

Table 2: Physicochemical characteristics of the anaerobic supernatant and the OFMSW FL for each experimental period (mean value \pm standard deviation)

Parameter	Units	Period 1	Period 2:	Period 2:
		Anaerobic supernatant	Anaerobic supernatant	OFMSW FL
Operation period	days	21-52	131-150	131-150
pH	-	7.82 \pm 0.07	7.87 \pm 0.08	4.14 \pm 0.08
TS	g/L	-	-	25.37 \pm 4.39
TVS	g/L	-	-	22.19 \pm 2.43
Conductivity	mS/cm	5.78 \pm 0.31	4.44 \pm 0.56	-
Alkalinity	mgCaCO ₃ /L	3346 \pm 375	2196 \pm 374	-
TSS	mg/L	145 \pm 9	33 \pm 1	-
COD	mg/L	134 \pm 11	31 \pm 5	62.22 \pm 5.37
sCOD	mg/L	97 \pm 4	26 \pm 5	31.33 \pm 3.09
TKN	mg/L	637 \pm 30	570 \pm 82	1128 \pm 319
NH ₄ -N	mg/L	593 \pm 28	559 \pm 75	315 \pm 62
NO ₃ -N	mg/L	0.4 \pm 0.1	0.8 \pm 0.1	< 0.2
NO ₂ -N	mg/L	0.5 \pm 0.1	0.4 \pm 0.1	<0.2
TP	mg/L	80 \pm 2	31 \pm 6	540 \pm 70
PO ₄ -P	mg/L	72 \pm 18	30 \pm 1	119 \pm 42

Nitrous oxide emissions

In the nitrification/denitrification process the formation of nitrate is inhibited. However, the three biochemical pathways that contribute to N₂O production can still occur since they are not associated with the activity of NOB. Furthermore, the implementation of an SBR is expected to lead to higher nitrous oxide emissions than an activated sludge process due to the fluctuation and higher nitrite concentrations in the mixed liquor that are observed during the reaction phases.

In period 1 the N₂O emissions from the treatment of the anaerobically co-digested supernatant were much lower than those of period 2 (*i.e.* 0.24% compared to 1.38% as per cent of influent nitrogen load). In period 2, the DO concentration during the aerobic reaction phase was lower. The DO level is a critical parameter affecting the N₂O emissions during nitrification. Low DO levels favour nitrous oxide emissions due to nitrifier denitrification [4]. Another parameter that is important is the nitrite level in the mixed liquor. In period 2 the higher vNLR and the sequence

change to anoxic/aerobic resulted in significant accumulation of nitrite which can also increase the emissions of N₂O.

The range of reported N₂O emissions from full scale WWTP is very wide and ranges from 0-14.6% of the nitrogen load for full scale WWTPs and 0-95% for lab scale investigations [3]. The treatment of sludge reject water is expected to result in higher nitrous oxide emissions than those of municipal wastewater treatment due to the higher nitrogen loads that are applied and potentially high nitrite/nitrate concentrations in the mixed liquor. The OFMSW and WAS co-digestion increases the nitrogen concentration in the resulting co-digestate therefore also increasing the potential N₂O emissions. As a result, the nitrous oxide emissions are higher than those reported in several works concerning the operation of activated sludge processes with BNR for municipal wastewater treatment [20-22]. Table 3 compares the nitrous oxide emissions obtained in this work with previous works for the treatment of sludge reject water. In our work, the strategy of implementing higher DO and process optimization for effective removal of nitrogen can decrease significantly the nitrous oxide emissions to low values. The use of OFMSW FL in combination with low DO and high vNLR could not mitigate the nitrous oxide emissions.

Table 3: N₂O emissions for the treatment of sludge reject water and anaerobic co-digestate

Wastewater	Process and conditions	N ₂ O emissions (% of N load)	Sampling details	Reference
Anaerobic supernatant from WAS & OFMSW co-digestion	Nitrification/denitrification High DO Low nitrite accumulation	0.24	On line Continuous 1 day	This study
Anaerobic supernatant from WAS & OFMSW co-digestion	Nitrification/denitrification Low DO High nitrite accumulation	1.38	On line Continuous 1 day	This study
Sludge reject water	2 stage partial nitrification – anammox	2.3	Off line Continuous 4 days	[3]
Sludge reject water	1 stage partial nitrification – anammox	1.3	Off line Grab samples 15 min/3×8 h	[23]
Sludge reject water	Nitrification	3.8	On line Continuous 12×6 h	[24]

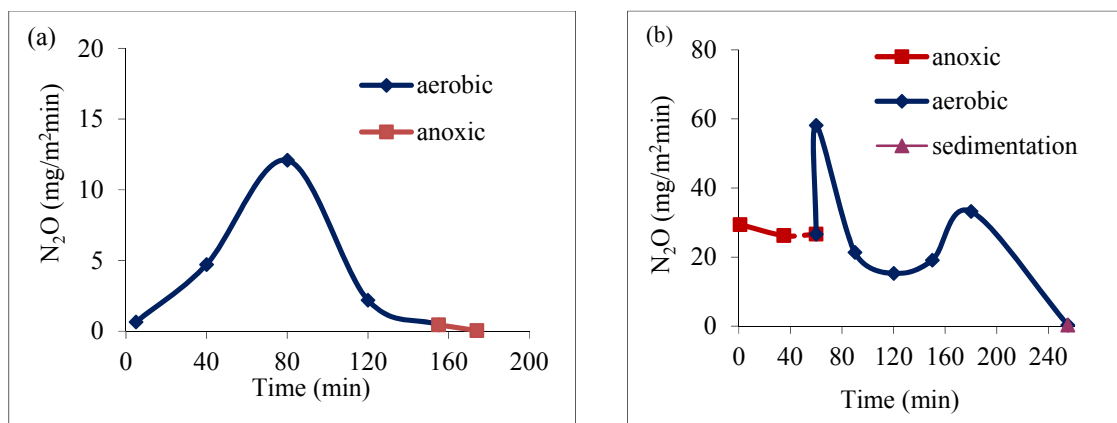


Figure 5: Variation of nitrous oxide emissions during the SBR operation for (a) period 1 and (b) period 2 respectively

Significant variation of the N₂O emissions was observed within the SBR cycle (Figures 5a & 5b). The aerobic reaction phase contributed much more to the N₂O emissions than the anoxic reaction phase. However, this does not mean that all the N₂O emitted during the aerobic phase is actually produced during aerobic conditions. It is also produced during the anoxic phase and can be subsequently stripped during the aerobic phase. The solubility of nitrous oxide in water is relatively high, so it may take some time for its stripping. This is characteristically represented with the peak that is observed at the beginning of the aerobic phase in Figure 5b. In period 2, the aerobic phase takes place just after the anoxic one, so the sudden introduction of air bubbles stripes off the N₂O that is produced during the anoxic phase. Since N₂O formed during the anoxic zone is stripped off during the aerobic zone it is very difficult to determine the individual contribution of nitrification and denitrification towards N₂O production.

Carbon dioxide, ammonia, methane and methyl mercaptan emissions

CO₂, NH₃, CH₄ and CH₃SH emissions from the SBR were also determined as a per cent of the influent nitrogen and COD load (Table 4). The CO₂ emissions were much higher during period 1 since the use of acetic acid resulted in lower biomass yield compared to period 2 in which OFMSW FL was added. The ammonia emissions were very low despite the fact that the mixed liquor contained significant free ammonia concentration (>2 mgN/L and often >4 mgN/L). An interesting finding is the significant levels of methane that were detected during period 2. In this period, the relatively low DO levels in the mixed liquor could have created local micro-anaerobic conditions within the sludge flocs, resulting in the production of gaseous emissions that are produced under anaerobic conditions such as methane. Furthermore, in period 2 fermentation liquid derived from OFMSW was added as carbon source rather than acetic acid, increasing the potential for methane emissions under micro-anaerobic conditions.

Table 4: Gaseous emissions measured in the SBR as % of the influent load

Parameter	Gaseous emissions in Period 1	Gaseous emissions in Period 2
N ₂ O (as % influent N load)	0.239	1.379
NH ₃ (as % influent N load)	0.0014	0.0173
CH ₄ (as % of influent COD load)	0.551	5.343
CO ₂ (as % of influent COD load)	55	31
CH ₃ SH (as % of influent COD load)	0.076	0.895

Mitigation strategy

This work showed that the nitrous oxide emissions can be decreased by (i) providing sufficient aeration during the nitrification stage so that the DO is maintained above 1.5 mg/L, (ii) applying a vNLR that is not higher than the system's nitrifying and denitrifying capacity. This way the accumulation of ammonium and nitrite is limited, (iii) apply the aerobic/anoxic sequence.

Conclusions

The SBR operation at DO= 1.5 mg/L and vNLR=0.81 kgN/m³d resulted in much lower nitrous oxide emissions (0.24% of influent nitrogen load) compared to the operation at lower DO (0.95 mg/L) and higher vNLR=1.08 kgN/m³d. Furthermore, the aerobic/anoxic SBR sequence resulted in much lower peak values of emissions than the anoxic/aerobic sequence. In the latter case the introduction of air just after the anoxic phase resulted in the stripping of the gaseous compounds formed during the anoxic phase. Operation at low DO and the addition of OFMSW fermentation liquid favoured emissions associated with anaerobic processes as micro-anaerobic conditions may prevail in the interior of sludge flocs.

Acknowledgments

This work was supported by the Marie Curie project entitled 'Low Environmental Footprint "Low Environmental Footprint Biological Treatment Processes for Waste and Wastewater Treatment", LEF-BIOWASTE, FP7-PEOPLE-2012-CIG, (Grant number 322333).

References

1. A. Mosier, C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger and O. van Cleemput, An overview of the revised 1996 IPCC guidelines for national greenhouse gas inventory methodology for nitrous oxide from agriculture, *Environ. Sci. Policy* 2(1999) 325-333.
2. IPCC, *Climate Change 2001: The scientific basis*, Cambridge University Press, Cambridge, 2001.
3. M.J. Kampschreur, H. Temmink, R. Kleerebezem, M.S.M. Jetten and M.C.M. van Loosdrecht, Review Nitrous oxide emission during wastewater treatment, *Water Res.* 43 (2009) 4093-4103.
4. J. Desloover, S.E. Vlaeminck, P. Clauwaert, W. Verstraete and N. Boon, Strategies to mitigate N₂O emissions from biological nitrogen removal systems, *Curr. Opin. Biotechnol.* , 23 (2012) 474 – 482.
5. J. Frijns, M. Mulder and J. Roorda, *Op weg naar een klimaatneutrale waterketen*. H₂O, 2008.
6. European Environment Agency, *Annual European Union greenhouse gas inventory 1990–2009 and inventory report 2011*.
7. J. López, G. Quijano, T.O. Souza, J. Estrada, R. Lebrero and R. Munõz, Biotechnologies for greenhouse gases (CH₄, N₂O, and CO₂) abatement: state of the art and challenges, *Appl. Microbiol. Biotechnol.* 97 (2013) 2277–2303.

8. J. Keller and K. Hartley, Greenhouse gas production in wastewater treatment: process selection is the major factor, *Water Sci. Technol.* 47 (2003) 43-48.
9. J. Desloover, H. De Clippeleir, P. Boeckx, G. Du Laing, J. Colson, W. Verstraete and S.E. Vlaeminck, Floc-based sequential partial nitrification and anammox at full scale with contrasting N₂O emissions, *Water Res.* 45 (2011) 2811-2821.
10. B.-J. Ni, M. Rusalleda, C. Pellicer-Nacher and B.F. Smets, Modeling nitrous oxide production during biological nitrogen removal via nitrification and denitrification: extensions to the general ASM models, *Environ. Sci. Technol.* 45 (2011) 7768-7776.
11. J.H. Ahn, S. Kim, H. Park, B. Rahm, K. Pagilla and K. Chandran, N₂O emissions from activated sludge processes, 2008-2009: results of a national monitoring survey in the United States, *Environ. Sci. Technol.* 44 (2010) 4505-4511.
12. J.H. Ahn, T. Kwan and K. Chandran, Comparison of partial and full nitrification processes applied for treating high-strength nitrogen wastewaters: microbial ecology through nitrous oxide production, *Environ. Sci. Technol.* 45 (2011) 2734-2740.
13. M.J. Kampschreur, W.R.L. van der Star, H.A. Wielders, J.W. Mulder, M.S.M. Jetten and M.C.M. van Loosdrecht, Dynamics of nitric oxide and nitrous oxide emission during full-scale reject water treatment, *Water Res.* 42 (2008) 812-826.
14. X.Y. Zhu and Y.G. Chen, Reduction of N₂O and NO generation in anaerobic-aerobic biological wastewater treatment process by using sludge alkaline fermentation liquid, *Environ. Sci. Technol.* 45 (2011) 2137-2143.
15. N. Frison, E. Katsou, S. Malamis, D. Bolzonella and F. Fatone, Biological nutrients removal via nitrite from the supernatant of anaerobic co-digestion using a pilot-scale sequencing batch reactor operating under transient conditions, *Chem. Eng. J.* 230 (2013) 595-604.
16. APHA AWWA WEF, Standard methods for the examination of water and wastewater, 20th ed.; American Public Health Association, American Water Works Association, Water Environment Federation, Washington, DC, 1998.
17. B. Eklund, Practical Guidance for Flux Chamber Measurements of Fugitive Volatile Organic Emission Rates, *J. Air Waste Manag. Assoc.* 42 (1992) 1583-1591.
18. S.G. Sommer, S.M. McGinn, X. Hao and F.J. Larney, Techniques for measuring gas emissions from a composting stockpile of cattle manure, *Atmospheric Environ.* 38 (2004) 4634-4652.
19. A. Chiumenti, F. Da Borso, T. Rodar and R. Chiumenti, Swine manure composting by means of experimental turning equipment, *Waste Manage.* 27 (2007) 1774-1782.
20. E. Sümer, A. Weiske, G. Benckiser and J.C.G. Ottow, Influence of environmental conditions on the amount of N₂O released from activated sludge in a domestic wastewater treatment plant, *Experientia* 51 (1995) 419-422.
21. Y. Kimochi, Y. Inamori, M. Mizuochi, K.Q. Xu and M. Matsumura, Nitrogen removal and N₂O emission in a full-scale domestic wastewater treatment plant with intermittent aeration, *J. Ferment. Bioeng.* 86 (1998) 202-206.
22. J. Sommer, A. Ciplak, E. Sumer, G. Benckiser and J.C.G. Ottow, Quantification of emitted and retained N₂O in a municipal wastewater treatment plant with activated sludge and nitrification-denitrification units, *Agrobiol. Res.* 51 (1998) 59-73.
23. N. Weissenbacher, I. Takacs, S. Murthy, M. Fuerhacker and B. Wett, Gaseous nitrogen and carbon emissions from a full-scale deammonification plant, *Water Environ. Res.* 82 (2010) 169-175.
24. D.J.I. Gustavsson and J.L. Jansen, Dynamics of nitrogen oxides emission from a full-scale sludge liquor treatment plant with nitrification, *Water Sci. Technol.* 63 (2011) 2838-2845.