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Development of an extended ASM3 model for predicting the nitrous oxide emissions in a full-scale wastewater treatment plant

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Abstract

An Activated Sludge Model #3 (ASM3)-based, pseudo-mechanistic model describing nitrous oxide (N\(_2\)O) production was created in this study to provide more insight into the dynamics of N\(_2\)O production, consumption, and emissions at a full-scale wastewater treatment plant (WWTP). N\(_2\)O emissions at the studied WWTP are monitored throughout the plant with an FT-IR analyzer, while the developed model encountered N\(_2\)O production in the biological reactors via both ammonia oxidizing bacteria (AOB) nitrification and heterotrophic denitrifiers. Additionally, the stripping of N\(_2\)O was included by applying a \(K_La\)-based approach that has not been widely used before. The objective was to extend the existing ASM3-based model of the plant and assess how well the full-scale emissions could be predicted with the selected model. The validity and applicability of the model were tested by comparing the simulation results with the comprehensive online data. The results show that the ASM3-based model can be successfully extended and applied to modelling
N$_2$O production and emissions at a full-scale WWTP. These results demonstrate that the biological reactor can explain most of the N$_2$O emissions at the plant, but a significant proportion of the liquid-phase N$_2$O is further transferred during the process.

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Supplementary information: Plant process characteristics, model process rate kinetics and parameters, goodness of fit –values and further modelling results are supplied as Supporting Information.

Introduction

Municipal wastewater treatment plants (WWTPs) have been recognized as potential sources of greenhouse gas (GHG) emissions$^1$. In particular, the research focus has been on nitrous oxide (N$_2$O), which is a significant GHG with approximately 300 times greater global warming potential than that of CO$_2$. N$_2$O is the most significant GHG emitted during wastewater treatment processes.$^2$ Additionally, it constitutes the most important ozone-depleting emission of the 21$^{\text{st}}$ century.$^3$ Given the significant environmental hazards posed by N$_2$O, mitigating N$_2$O emissions is of utmost importance and will greatly impact the sustainability of WWTPs.

In WWTPs, N$_2$O can be emitted during the biological nitrogen removal processes, including during autotrophic nitrification, autotrophic denitrification, and heterotrophic denitrification, which have been identified as the major emission sources in wastewater treatment. Due to the challenges of directly monitoring N$_2$O emissions at WWTPs, N$_2$O emissions are often estimated by applying a fixed standard emission factor. This approach has several limitations: the fixed emission factor is not only inconsistent with respect to recent studies, which show great variation in the reported N$_2$O emissions from individual WWTPs$^{2,4-5}$, but it might also lead to underestimating the real level of
emissions. Furthermore, the fixed emission factor does not take into account the impact of the different process conditions and configurations on N\textsubscript{2}O emissions, which assumedly affect the ultimate N\textsubscript{2}O production rates significantly.

With improved instrumentation and long-term monitoring, there is a growing interest in the mathematical modelling of N\textsubscript{2}O production in WWTPs. This would enable plant-specific estimations of N\textsubscript{2}O production, since such modelling would take into account each individual plant’s layout, process configuration, and operational characteristics. Furthermore, a comprehensive mathematical N\textsubscript{2}O model can become a powerful tool for developing new N\textsubscript{2}O-emission mitigation strategies at WWTPs.

Currently, there are several different models available for estimating N\textsubscript{2}O production. While all the models are based on the known metabolic pathways of N\textsubscript{2}O production, there is still much variety in their structure depending on the number of pathways included, stoichiometry, and kinetics. The single-pathway models describe N\textsubscript{2}O production through one of the AOB reactions (i.e., either AOB denitrification or NH\textsubscript{2}OH oxidation), whereas the two-pathway model employs both of the production routes. Since the two-pathway models increase the number of parameters included, leading to laborious calibrations, the single-pathway models are potentially more convenient in certain environments, such as environments with high dissolved oxygen (DO) concentrations (> 1.5 g · m\textsuperscript{-3}). Several comprehensive reviews of the current N\textsubscript{2}O models and general GHG models that include N\textsubscript{2}O production have been published. While modelling has proved to be a potentially applicable tool for estimating N\textsubscript{2}O production and emissions at WWTPs, there is still a need for further research because only a limited number of studies have been conducted on full-scale WWTPs using comprehensive online data. Previously, a few
Activated Sludge Model (ASM)-type, single-pathway models have been applied to modelling full-scale WWTPs \(^{10-11}\), but in none of the previous publications an Activated Sludge Model #3 (ASM3)-based model has been used. The ASM3 provides more accurate reaction kinetics \(^{12}\) and allows for easier calibration. It is also commonly used for modelling not only within the field of wastewater engineering and related disciplines, but also by stakeholders as an inherent part in WWTP design and operation.

The modelling of \(\text{N}_2\text{O}\) emissions and related monitoring data campaigns have traditionally focused solely on biological reactors. Thus, there is no clear understanding of how well the models can explain the total \(\text{N}_2\text{O}\) emissions on a plant-wide scale. Moreover, there is a clear lack of knowledge regarding the contribution of post-aeration processes to \(\text{N}_2\text{O}\) dynamics. On a practical level, it is challenging to determine the total \(\text{N}_2\text{O}\) emissions at open-ditch plants, whereas underground plants are more convenient for monitoring \(\text{N}_2\text{O}\) emissions on a plant-wide scale. Furthermore, in order to provide more insights into \(\text{N}_2\text{O}\) dynamics, the stripping of \(\text{N}_2\text{O}\) should be considered for both the gaseous and liquid phase. Generally, the stripping of \(\text{N}_2\text{O}\) has neither been widely discussed nor mentioned in the literature with regard to the modelling of \(\text{N}_2\text{O}\) production and emissions, even though it has a crucial impact on the modelled \(\text{N}_2\text{O}\) emissions as well as the \(\text{N}_2\text{O}\) production model. Moreover, from an environmental perspective the liquid-phase \(\text{N}_2\text{O}\) has not traditionally been seen as being as significant as the gaseous emissions \(^{13}\), even though the contribution of the liquid-phase \(\text{N}_2\text{O}\), if emitted later on, can be significant in terms of the overall carbon footprint of the plant \(^{14}\).

The objective of this study was to create an ASM3-based, pseudo-mechanistic \(\text{N}_2\text{O}\) model describing \(\text{N}_2\text{O}\) production via AOB and heterotrophic denitrifiers at a full-scale WWTP. Moreover, the stripping of \(\text{N}_2\text{O}\) was included and an attempt was made to identify the main sources
and sinks of N₂O in the treatment process. The goal was to create a tool for aiding a plant’s process control, and thus, the model was created to be ASM3 compatible and kept as simple as possible. The validity and applicability of the model were tested by comparing the simulation results with the comprehensive monitoring online data collected from the underground plant of Viikinmäki WWTP. Furthermore, the capability of the model to explain the plant-wide emissions, and thus, the sufficiency of the selected modelling approach, was assessed.

Material and methods

Field site description

Viikinmäki WWTP is the largest wastewater treatment facility in the Nordic countries, with a population equivalent to 1,100,000 p.e. The WWTP treats wastewater from the metropolitan area of Helsinki, accounting for a total of up to 800,000 inhabitants. The plant also receives industrial wastewater, which accounts for approximately 7% of the total inflow. The average influent concentrations and operation are presented in table 1 as supplementary information (SI). In this underground plant, the air from the process tunnels is conducted outside via an exhaust air channel.

The treatment process consists of influent pumping, screening, grit removal, primary sedimentation, and biological treatment. Biological treatment at the Viikinmäki WWTP is based on an activated sludge process (ASP) and denitrifying post-filtration. The plant has altogether nine identically configured ASP lines, one of which was studied in this work. Each biological reactor in the ASP is divided into six zones, including one anoxic pre-denitrifying zone, two alternating switch zones, and three aerated nitrifying zones. Additionally, each biological reactor is equipped with a mixing zone and a degassing zone (fig. 1).
Adjustments to the aeration process and control of the switch zones in each ASP lines are performed based on online ammonium measurements at the end of the biological reactor. Under normal process conditions, three or four of the six zones are aerated. The ASP is normally run at a DO set point of 3.5 g O$_2$·m$^{-3}$ using PI-control for each of the aerated zones.

![Diagram](image)

**Figure 1.** A schematic representation of the ASP and the locations of the online analyzers. Mixing zone = 500 m$^3$, Z$_1$ = 1500 m$^3$, Z$_2$ to Z$_6$ = 1900 m$^3$, degassing zone = 385 m$^3$, S$_1$ and S$_2$ = 6700 m$^3$. $Q_{in}$ = Influent flow, SS = Suspended solids, NH$_4$-N = Ammonia nitrogen, $Q_{air}$ = Air flow, DO = Dissolved oxygen, $N_2O_{aq}$ = Dissolved nitrous oxide, $N_2O_g$ = Gaseous nitrous oxide, $Q_{RAS}$ = Return activated sludge, $Q_{WAS}$ = Waste activated sludge, NO$_3$-N = Nitrate nitrogen, MLSS = Mixed liquid suspended solids, Alk = Alkalinity.

**Process monitoring**

The WWTP is fully automated and monitored via online analyzers. The most important analyzers used in this study are listed in table 2 as SI. The locations of the online analyzers at the studied ASP line are presented in figure 1.

In addition to online process monitoring, the plant is monitored via laboratory analysis. The most important analyses used in this study are presented in table 3 as SI. The samples
were collected from influent, mechanically treated water, secondary clarified water, and effluent water.

**N₂O analyzer equipment**

*FT-IR analyzer*

The plant’s total N₂O emissions have been continuously measured and monitored since 2012 via a Gasmet CEMS II system situated in the exhaust air channel ². Since the WWTP was built underground, the measurements cover the gaseous N₂O emissions of the whole biological process, including denitrifying post-filtration. The Gasmet CEMS II system—which consists of an online Fourier transform infrared (FT-IR) analyzing unit, an industrial computer unit, and a sampling system—is connected to the plant’s automation system, which stores the data in the WWTP’s internal database.

**N₂O sensors**

The fourth and the sixth zones of the aeration tank were equipped with an online, Clark-type, micro sensor (Unisense A/S, Denmark) placed directly in the tank. The sensors continuously measured the dissolved N₂O concentrations during the liquid phase with a measurement range of 0.1 – 500 µM. The liquid-phase N₂O in zone 4 and zone 6 was monitored during two measurement campaigns consisting of 12 days and 5 days, respectively, in April 2017. Zone 4 was the first continuously aerated zone in the ASP, and it was selected in order to measure N₂O production during aeration without significant accumulation from the previous zones. Zone 6 was the last aerated zone under study, and it was selected in order to validate the dynamics of liquid-phase N₂O in the aeration reactor by capturing the variation in liquid-phase N₂O in two zones simultaneously.
The sensors were two-point calibrated prior to the measurement campaigns. Additionally, the
switching zone 3 was monitored using the on-line micro-sensor during another measurement
campaign for 14 days (22.3–3.4.2017) in order to capture N₂O production during the anoxic and
aerated phases prior to zone 4.

In order to estimate the liquid-phase N₂O concentrations after the aeration basin,
hourly grab samples were taken from the plant’s effluent and from the secondary clarifier (average
retention time: 8.5 h) influent and effluent during a six-hour period (9 a.m. – 3 p.m.) in a two-day
measurement campaign in 2016 and another measurement campaign in 2017. The samples were
collected in a lidded container (0.01 m³) and measured immediately using a Unisense N₂O sensor.

The model’s description

The proposed extended ASM3 model considers, in addition to the original ASM3 processes, N₂O
production through both the NH₂OH oxidation pathway via AOB and heterotrophic denitrification.
The NH₂OH modelling approach for N₂O production was assumed to be suitable due to the
dynamic and relatively high DO concentrations (1.5 – 3.8 g O₂ · m⁻³) in the aerated zones of the
ASP and the low concentrations of NO₂⁻ (0.1 and 0.7 g N · m⁻³)8-9. The SI in table 4 summarizes
the full list of state variables included in the extended ASM3 model.

The model was created stepwise by first extending the original nitrogen conversion
processes in the ASM3 into two-step nitrification and two-step denitrification reactions. Following
this, denitrification was extended into a four-step denitrification reaction and the NH₂OH pathway
was implemented. The stepwise approach was used in order to better identify the mistakes during
each step. The extension of the ASM3 to describe the two-step nitrification and two-step
denitrification reactions was done according to specifications by15.
The assumptions for the four-step denitrification reaction and N\textsubscript{2}O production via heterotrophic denitrification are based on the Activated Sludge Model for Nitrogen (ASMN)\textsuperscript{16}. This approach was chosen because it can be assumed that the denitrification reaction at the WWTP is not carbon limited. Additionally, there is limited knowledge on carbon oxidation and nitrogen reduction kinetics; such knowledge is required for the other denitrification modelling approaches, such as the indirect coupling approach\textsuperscript{8,17-18}. The ASMN is based on ASM1, and therefore it did not originally include the storage reactions, anoxic endogenous respiration, and anoxic respiration of the storage substances that were additionally extended in this work for the ASM3 (processes p3a – c, p5a – c, p7a – c, p9a – c in the SI for table 5). Here, the two-step denitrification reaction\textsuperscript{15} was further extended into a four-step denitrification reaction according to the reaction kinetics and stoichiometry described in the ASMN. Additionally, the description of terms in the ASMN was done according to specifications by\textsuperscript{19}.

The assumptions for N\textsubscript{2}O production through the NH\textsubscript{2}OH pathway are based on a previous single-pathway model (Ni et al., 2013b) corresponding to four new biological reactions (presented in table 5 as SI: p10a1 – p10a4, SI). A few modifications were made to the original model: (i) the bacteria yield was added in p10a1 and p10a2 (table 5 as SI), (ii) the N\textsubscript{2}O production (p10a4) kinetics employ a reaction-specific growth rate instead of a reduction factor, and (iii) the AOB growth was attributed to the p10a2 process. The p10a4 process describes N\textsubscript{2}O production via the NH\textsubscript{2}OH pathway. In the model, it is assumed that N\textsubscript{2}O is produced during the reduction of NO, which is in turn produced during the oxidation of NH\textsubscript{2}OH. During the production of N\textsubscript{2}O, NH\textsubscript{2}OH is also consumed as an electron donor.

Since NO is an intermediate in both of the described N\textsubscript{2}O production pathways, the component was divided into NO\textsubscript{AOB} and NO\textsubscript{DEN} by adding two separate state variables attributing
the AOB-related production of NO and heterotrophic denitrification-producing NO, respectively. This made it possible to prevent the NH$_2$OH pathway reaction during heterotrophic denitrification, and vice versa. In the model, the NH$_2$OH oxidation pathway for N$_2$O production does not require DO, thus theoretically part of the NO produced during denitrification could have been utilized by the AOB via the NH$_2$OH pathway under anoxic conditions.

A comprehensive modelling study that includes data quality proofing, influent fractionation, and model construction was already done at the plant. The influent fractions were created based on the previous findings according to the online analyzers’ data and laboratory results collected from the ASP influent channel (table 3 as SI). The variation in the soluble components was assumed to follow the pattern of the influent NH$_4$-N, while the variation in the particulate matter was assumed to follow the pattern of the SS measurement. Phosphorus was implemented in the influent model as a constant value since it does not have an impact on the N$_2$O dynamics in wastewater treatment.

**N$_2$O stripping**

Prior studies have shown that aerated zones in the ASP are the primary emission sources of N$_2$O, since the liquid-phase N$_2$O is mainly stripped during the aeration phase. The stripping of N$_2$O in each of the zones was estimated using a mathematical approach based on the modelled N$_2$O concentrations ($S_{N_2O}$) and air flow ($Q_A$) in the aerated zones. Equation 1.1 is as follows:

$$R_{N_2O} = H_{N_2O}^p c \cdot S_{N_2O} \left( 1 - e^{-\frac{K_{L,A} N_2O V_R}{H_{N_2O}^p c Q_A}} \right) \cdot \frac{Q_A}{V_R}, \quad (1.1)$$

where $R_{N_2O}$ is the stripped N$_2$O, that is to say, the emission rate of N$_2$O (g m$^{-3}$ d$^{-1}$), $H_{N_2O}^p c$ is the Henry’s constant (mol · L$^{-1}$ · bar$^{-1}$) at the process temperature, which was calculated according to
$K_La_{N_2O}$ is the mass transfer coefficient for $N_2O$ at the process temperature, which was calculated according to $^4$ for each zone separately, $V_R$ is the volume of the aerated zone ($m^3$), and $Q_A$ is the air flow in the zone ($m^3 \cdot d^{-1}$).

The stripping from the anoxic zones was first calculated based on an approach proposed by $^{22}$, but it proved to be negligible compared to the aerated zones as it was only less than 1.5% of the total emissions from the aerated zones. This was due to the low stripping effect of the non-aerated zones compared to the aerated zones. Additionally, the measured liquid-phase concentrations of $N_2O$ in the anoxic zones were smaller compared to those in the aerated zones. For the sake of simplicity, the stripping of $N_2O$ from the anoxic zones was not included in the model.

The measured $N_2O$ emissions that cover the emissions of the whole plant were scaled in order to compare the measured and modelled $N_2O$ of the studied ASP line. The nine identically configured ASP lines were assumed to contribute evenly in relation to the inflow ($m^3 \cdot d^{-1}$) to the measured $N_2O$ emissions in the exhaust air channel. The contribution of the studied ASP line to the total $N_2O$ emissions was estimated based on the percentage of the total wastewater treated in the ASP line, which is on average 10–12%. The total air flow ($m^3 \cdot s^{-1}$) through the exhaust air channel was estimated based on the plant’s ventilation ($100 \pm 10 \ m^3 \cdot s^{-1}$) and used for estimating the hourly emissions (kg $N_2O$-N $\cdot h^{-1}$).

**Model calibration and validation**

The model simulation was performed using GPS-X (version 6.5). The ASP was modelled using a tanks-in-series model with eight zones (a mixing zone, anoxic pre-denitrifying zones, and aerated
nitrifying zones, accounting altogether for six reactors and a degassing zone) and two parallel secondary clarifiers (fig. 1). The denitrifying post-filtration was excluded from the model because its N₂O emissions have been observed as insignificant. According to good modeling practice, a steady state simulation was conducted prior to every further simulation of the desired period of time.

The model was calibrated using the comprehensive online data collected from the plant (table 2 as SI). The model was validated and tested for a period of five days, which was done together with another monitoring data campaign at the plant (table 2 as SI). The process conditions during the calibration and validation phases were stable. Since the aerated zones are automatically controlled and the influent load to the plant varies constantly, the aeration periods differed slightly. During the calibration (8–19.4.2017) phase, zones 4–6 were continuously aerated, the switching zone 3 was aerated twelve times for two to three hours, and zone 1 was anoxic. During the validation (25–30.4.2017) phase, zones 4–6 were continuously aerated, the switching zone 3 was aerated four times for periods of 2–4 hours, and zone 1 was anoxic.

Parameter estimation and goodness-of-fit evaluation

Given the rising number of kinetic parameters that derived from the performed extension steps, calibration of the final extended ASM3 model was challenging. Therefore, parameter estimation was performed following the specifications by, while a detailed description for a specific modelling case can also be found in a study by. The proposed approach examines the model’s kinetic parameter correlation and is based on constructing a confidence region that encompasses the likelihood confidence region. The estimation of the single kinetic parameters (θ) was done using the residual sum of squares (RSS) objective function J(θ) (equation 1.2):
where $N$ is the number of measurements, $y_i$ the measured data, and $\hat{y}_i$ the outputs predicted by the model.

Calculation of the confidence regions for the optimized parameters was performed using the following equation 1.3:

$$SSR(\theta) \leq SSR(\theta_0) \cdot \left(1 + \frac{p}{n-p} \cdot F_{p,n-p}^{\alpha}\right).$$

(1.3)

Here, $SSR(\theta_0)$ refers to the minimum sum of the squared residuals, $p$ refers to the number of parameters, $n$ refers to the number of measured values, and $F_{p,n-p}^{\alpha}$ refers to the F value taken from the F-distribution for a confidence interval $\alpha = 0.05$ with $p$ and $p-n$ degrees of freedom.

For the present study, we used $F_{2,\infty}^{0.05} = 2.996$, which is also in accordance with that used by 28.

The increasing complexity of the model not only demands appropriate parameter estimation, but also goodness-of-fit measures to evaluate the quality of the model. Therefore, the modified Nash-Sutcliffe coefficient ($E_j$) was determined using equation 1.4, where $M_i$ refers to the measured data, $E_i$ to the modelled data, and $M_{\bar{0}}$ to the mean of the measured data.

$$E_j = 1 - \frac{\sum_{i=1}^{N} |M_i - E_i|}{\sum_{i=1}^{N} |M_i - M_{\bar{0}}|}$$

(1.4)

As $E_j$ can be sensitive to extreme values, the modified version applies absolute values to reduce the effect of the squared terms. A Nash-Sutcliffe coefficient of 1 corresponds to a perfect match of the modelled and measured data, whereas a coefficient of 0 indicates that the models prediction is as accurate as the mean of the measured data. 30 A coefficient below 0 refers to a simulation worse than the average of the measured data.
Additionally, the Root Mean Square Error (RMSE) was calculated for selected parameters using equation 1.5, where \( n \) represents the number of measured values, \( y_i \) the modelled values, and \( \hat{y}_i \) the measured values. The smaller the RMSE value, the closer the modelled and measured values.

\[
RMSE = \sqrt{\frac{1}{n} \sum_{i=1}^{n} (y_i - \hat{y}_i)^2}
\]  

(1.5)

Results

The liquid \( \text{N}_2\text{O} \) monitoring results

The average liquid-phase \( \text{N}_2\text{O} \) concentration in the aerated zone 4 was 0.04 g N · m\(^{-3}\) and 0.03 g N · m\(^{-3}\), whereas it was 0.09 g N · m\(^{-3}\) and 0.11 g N · m\(^{-3}\) in the aerated zone 6 during the calibration and validation phases (fig. 3). An online measurement campaign in the switching zone 3 showed that during the anoxic periods, the average \( \text{N}_2\text{O} \) concentration in zone 3 was 0.027 g N · m\(^{-3}\), whereas we observed peak concentrations of up to 0.16 g N · m\(^{-3}\) during or directly following the aerated periods. The online and grab-sample measurements at the WWTP show that liquid-phase \( \text{N}_2\text{O} \) accumulated in the ASP and that the measured \( \text{N}_2\text{O} \) concentrations were notably higher in the aerobic environments compared to the anoxic environments.

We collected the hourly grab sample measurements from the influent and effluent of the secondary clarifiers and the plant effluent. The measured values prior to the secondary clarification varied between 0.38 - 0.43 g N · m\(^{-3}\), 0.50 - 0.59 g N · m\(^{-3}\), and 0.29 - 0.31 g N · m\(^{-3}\) during the first, second, and third measurement campaigns. The measured values in the effluent of the secondary clarification varied between 0.31 - 0.36 g N · m\(^{-3}\), 0.35 - 0.49 g N · m\(^{-3}\), and 0.32 - 0.34 g N · m\(^{-3}\) during the first, second, and third measurement campaigns. We simulated
similar concentrations of liquid-phase N\textsubscript{2}O in the secondary clarification effluent in the model, with concentrations of 0.32 g N \cdot m\textsuperscript{-3} and 0.30 g N \cdot m\textsuperscript{-3} during the calibration and validation phases, respectively.

This indicates that a notable proportion of the N\textsubscript{2}O produced in the aeration reactor is further conveyed from the ASP. Additionally, part of the N\textsubscript{2}O can be conveyed from the secondary clarifier back to the mixing zone of the biological reactor along with return activated sludge (RAS) and internal sludge recycling.

During the first measurement campaign, we did not detect N\textsubscript{2}O in the effluent of the plant except for in one sample measurement of 0.01 g N \cdot m\textsuperscript{-3}. Consistently, during the second and the third measurement campaigns the detected effluent concentrations were low, varying between 0.00 - 0.05 g N \cdot m\textsuperscript{-3}.

Parameter estimation

The kinetics and stoichiometric parameter set (table 6 as SI) consisted of 63 parameters with 23 stoichiometric and 40 kinetic parameters. We carried out the calibration procedure based on the knowledge acquired about the process. We chose a match between the simulated and online data in the SI for table 2 as the main criteria for defining the quality of the model prediction in terms of the nitrogen conversion reactions and kinetics —that is to say, nitrification, N\textsubscript{2}O production, and denitrification—using the monitoring data (figure 1, table 2, SI).

The task began with us calibrating the ammonia oxidation (\(\mu_{\text{AOB,AMO}}\)). In addition, as our particular interest was on N\textsubscript{2}O production, we adjusted the \(\mu_{\text{AOB,NOR}}\) until the modelling results were in accordance with the measured values, prioritizing the dynamics of the production of N\textsubscript{2}O. We chose a value for the saturation coefficient of NH\textsubscript{2}OH (\(K_{\text{AOB,NH2OH}}\)) that was the same
as the one for the saturation coefficient of nitrate and nitrite. We kept the remaining stoichiometric
and kinetic parameters at the values given in the literature (table 6, SI).

We found that the calibrated value of $\mu_{\text{AOB,AMO}} = 1.21 \, \text{d}^{-1}$ was satisfactory with
regard to the joint 95% confidence regions and also in line with the range given in literature, which
varies between 0.768 and 5.184 $\, \text{d}^{-1}$\footnote{1}. As exemplarily for the $S_{\text{NO}}$ (shown in figure 2), the
parameter estimation approach revealed a high number of possible parameter combinations. In this
study, the model employs a maximum growth rate for the NOR-mediated reaction, that is to say,
it employs the NH$_2$OH oxidation pathway instead of an anoxic reduction factor and the HAO-
mediated reaction rate, as discussed in a study by\textsuperscript{10}. The calibrated value of 0.15 $\, \text{d}^{-1}$ found for the
maximum NOR-mediated reaction rate is, however, equivalent to the range of the values presented
by\textsuperscript{10}.

The original ASMN is based on ASM1 processes, and in this study it was extended
for ASM3, leading to seven new parameters in total. For the sake of simplicity, we applied the
same yield coefficient values ($Y_{\text{H,NO}}, Y_{\text{H,N2O}}$) for heterotrophic growth and for storage ($X_{\text{STO}}$) in
anoxic growth ($Y_{\text{STO,NO}}, Y_{\text{STO,N2O}}$) for all of the denitrifying reactions. Additionally, we used the
reduction factor values presented by\textsuperscript{15} for NO$_2^-$ reduction under anoxic conditions ($\eta_{\text{H,endo,NO}}$)
(0.35 ... 0.70) for calibrating the NO reduction and N$_2$O reduction values ($\eta_{\text{H,endo,NO}}, \eta_{\text{H,endo,N2O}}$).
Figure 2. Joint 95% confidence region for the maximum AMO-mediated reaction rate for NH$_4^+\text{-NH}_2\text{OH}$ ($\mu_{\text{AOB,AMO}}$) and the saturation/inhibition coefficient for O$_2$ in NH$_4$ oxidation ($K_{\text{AOB,O}_2,\text{NH}_4}$) in relation to the dissolved nitrous oxide concentration $S_{\text{N}_2\text{O}}$.

**Modelling results**

The model was able to reproduce the variation in the measured liquid-phase N$_2$O concentrations during the calibration and validation phases in zones 4 and 6, the zones that were continuously aerated (fig. 3). Likewise, the model was able to reproduce the variation in N$_2$O in zone 6 more precisely than in zone 4. This could be due to the fact that the switching zone 3, while it is aerated, affects N$_2$O production in zone 4, which could not be fully captured by our model. Furthermore, the total concentrations of N$_2$O in zone 4 were lower than in zone 6, which means that the measured values and the modelling results for zone 4 were more sensitive to all kinds of variations in the full-scale WWTP. Altogether, the model was able to capture the dynamics of the liquid-phase N$_2$O in the aeration tank, which is demonstrated by the two simultaneous measurements in two different
zones and additionally by the grab sample measurements in the secondary clarification influent and effluent.

Figure 3. Modelled (dashed line) and measured (solid line) $N_2O$-N concentrations in the liquid in the continuously aerated zones 4 (above) and 6 (below) during calibration (left) and validation (right).

The model was also able to capture the dynamics of the measured $N_2O$ emissions in the exhaust air channel (fig. 4). However, while the model was able to predict the variations in $N_2O$ emissions, the basic level of the modelled values was notably higher compared to the measured emissions.
The ammonia oxidation is directly linked to all of the main nitrogen conversion reactions in the ASP. The model was able to reproduce the measured NH$_4^+$ peaks in the effluent of the ASP throughout the simulation (fig. 1 as SI). During the calibration period, the modelled peaks were able to reach the measured values throughout the simulation. Consistently, the model was able to reproduce the measured peaks during the validation phase.

Additionally, the model was able to reproduce most of the measured NO$_3^-$ variation in the effluent of the ASP (fig. 2 as SI). During both the calibration and validation phases, the model was not able to simulate the variation during weekend periods (days 10–12 during calibration and days 3–5 during validation), which can be explained by the changes in the characteristics of the influent since the proportion of industrial wastewater is smaller. The NO$_3^-$ variation is directly linked to denitrification, and therefore, to variations in the organic load of the plant. Overall, estimating the diurnal variations in the organic load is difficult at full-scale plants, which is reflected in our modelling of the NO$_3^-$ concentrations. Additional calibration results for
the ASP, such as MLSS in the biological reactor and TSS in the ASP influent, are presented in the SI. The results of the Nash-Sutcliffe coefficients and RMSE are presented in the SI of table 7.

Discussion

The extended ASM3-based model included N$_2$O production through the NH$_2$OH oxidation pathway and heterotrophic denitrification pathway in addition to the original ASM3 processes. The model was able to capture the measured N$_2$O variation and dynamics during both liquid and gaseous phases at the plant. The model was able to reproduce most of the variation in the liquid-phase N$_2$O concentration in the continuously aerated zones 4 and 6 during the calibration and validation phases. These modelled results are also supported by previous NH$_2$OH oxidation pathway modelling approaches, as the model was able to capture the N$_2$O production dynamics in accordance with the DO dynamics $^8,^{11}$.

These modelling results are in accordance with the measured N$_2$O concentrations and previous studies, suggesting that most N$_2$O production takes place in the aerated zones of the ASP $^{21}$. The previous N$_2$O emission study done at Viikinmäki WWTP demonstrated a correlation between the number of zones aerated and N$_2$O emissions $^2$, which could be also related to increased AOB-related N$_2$O production due to increased aeration. However, the correlation can also be related to an increased ammonium load, which results in the need for an increased aeration volume. Altogether, given the low N$_2$O concentration in the anoxic zones of the ASP due to the N$_2$O mainly being consumed by denitrification and the notably smaller stripping effect compared to the aerated zones, we can assume that the contribution of the anoxic zones to the plant’s gaseous-phase N$_2$O emission is negligible.
To the best of the authors’ knowledge, this study is the first of its kind to present the dynamics of liquid-phase N$_2$O using two simultaneous measurements in the aeration reactor. This demonstrates that the model can capture the dynamics of liquid-phase N$_2$O in the aeration process, and therefore, the model can be seen as a possible tool for studying the effects of different operational strategies on the mitigation of N$_2$O at the plant.

The modelled emissions during both the calibration and validation phases were relatively consistent with the dynamics of the measured N$_2$O emissions collected from the exhaust air channel. It is important to note here that the measured gaseous emissions represent the total N$_2$O emissions for the whole plant. Thus, in order to compare the modelled and measured N$_2$O emissions, we scaled the measurement for the studied treatment line. For instance, the model predicted slightly stronger fluctuations compared to the measured emissions, possibly because the fluctuations in measured emissions are more stable due to the contribution of all the ASP lines. Nevertheless, the modelling results suggest that the biological model is able to predict most of the variations in the emissions and that the most significant part of the emissions is produced in the aeration reactor.

While the model was able to capture the variation in the plant’s N$_2$O emissions, the basic level of the modelled emissions was notably higher compared to the measured values during both the calibration and validation phases. The level of the modelled emissions could not be fixed via model calibration, that is, it could not be fixed by calibrating the N$_2$O production in the liquid phase. Therefore, an important concern demonstrated in this study is that the applied stripping model seems to over-estimate the amount of stripped N$_2$O, at least during certain periods, which leads to an over-estimation of the total emissions. Interestingly, another simulation from late summer 2016 showed a better fit for gaseous-phase emissions suggesting that the observed large
variations in emissions are caused by changes in stripping, not necessarily in N₂O production. While the stripping model has a crucial impact on how the total emissions of N₂O are modelled, there has not been any discussion in the literature regarding the stripping models used in the previous N₂O modelling studies.

The grab sample measurements prior to the secondary clarification showed an average concentration of 0.40 g N · m⁻³, 0.55 g N · m⁻³, and 0.30 g N · m⁻³ during the first, second, and third measurement campaigns, respectively. Similarly, for instance reported liquid-phase N₂O concentrations of up to 0.3 g · m⁻³ in the secondary clarification. Based on the measurements in the secondary clarification influent and effluent, we estimated that 10–50% of the N₂O entering the secondary clarification could potentially be emitted or consumed during the process, possibly due to the stripping of N₂O or else, to a smaller extent in this case, to heterotrophic denitrification leading to the generation of N₂. While there is a lack of research regarding the N₂O emissions originating from the secondary clarification, these findings are supported by the results provided by which demonstrate that more than 30% of the N₂O emissions can be emitted from the secondary clarifiers. In this case, significant stripping in the secondary clarifiers in relation to the anoxic zones is supported by the fact that the aeration tank is quite deep (12 m) compared to the clarifiers and very little sludge is stored in the secondary clarifiers. Overall, the contribution of secondary clarification to the dynamics of N₂O production and consumption should be further studied.

Nevertheless, the liquid-phase N₂O concentrations in the effluent of secondary clarification suggest that the major proportion of the liquid-phase N₂O is further conveyed to post-denitrifying filtration. Since the measured N₂O concentrations in the plant effluent were insignificant, we can assume that the N₂O is consumed during denitrifying post-filtration.
Moreover, the measured and modelled concentrations of liquid-phase N\textsubscript{2}O in the secondary clarification demonstrate that an important proportion of the N\textsubscript{2}O may not be stripped out during the aerated zones, unlike current assumptions suggest. The fate of this potential source of N\textsubscript{2}O emissions later on in the process in the air or in the receiving water body should be further studied. Further research would be required in order to investigate the mitigation of emissions via denitrification. This modelling approach, including the stripping of N\textsubscript{2}O, might also provide a useful tool to investigate whether the mitigation methods at the plant should be implemented during the process operation in order to restrict the production of N\textsubscript{2}O or the stripping of N\textsubscript{2}O during aeration.

While the results show that the NH\textsubscript{2}OH oxidation pathway model was able to describe N\textsubscript{2}O production at the studied Viikinmäki WWTP, it should be noted that the model may not be able to reproduce all N\textsubscript{2}O production at other WWTPs. The NH\textsubscript{2}OH oxidation pathway models are reportedly applicable for environments with a relatively high DO concentration\textsuperscript{9-10}, whereas these models have not been able to predict N\textsubscript{2}O production in environments with NO\textsubscript{2}\textsuperscript{-} accumulation\textsuperscript{11}. Furthermore, the results presented in this study do not necessarily rule out the possibility that the AOB denitrification could contribute to the ultimate N\textsubscript{2}O emissions at the Viikinmäki WWTP. This N\textsubscript{2}O production pathway, which is also known as nitrifier denitrification, implies the reduction of NO\textsubscript{2}\textsuperscript{-} to NO and further to N\textsubscript{2}O. According to current understanding, the pathway is favored during nitrification at low DO concentrations. A full investigation of the pathways that are dominantly contributing to the N\textsubscript{2}O pool will require stable nitrogen and oxygen isotope analysis, which will be likely addressed in future research studies.

Nevertheless, in this study we extended the ASM3 for the first time by modelling N\textsubscript{2}O production at a WWTP. The results show that an ASM3-based model can be successfully
extended and applied to the modelling of N₂O production at a full-scale WWTP, as the model was able to produce not only the measured N₂O production but also the comprehensive online monitoring data collected at the plant. This study also introduced N₂O stripping into the process dynamics, which had been lacking from previous models. Overall, this is an important step towards applying N₂O models to a full-scale WWTP.

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