



POWERSTEP

WP4: Nitrogen management in side stream

D4.1: Experience and performance data for the implementation of nitrification on a two-stage WWTP



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Abstract	Analysis of optimal performance factors and key figures for nitrification as sludge age, feeding rate, oxygen demand or temperature dependent on local circumstances. The deliverable is an experience report on qualitative and quantitative description of challenges, problems and appropriate solutions for a stable nitrification process.

Dissemination level of this document

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Executive Summary

The project POWERSTEP is conceived to achieve a real paradigm shift in wastewater treatment processes: the overarching goal is to convert sewage treatment plants (STEPS) in power production facilities (POWER) while still achieving a high effluent quality of the treated wastewater.

Work package 4 deals with the biological or physical nitrogen treatment in sidestream. The nitrogen in the sludge dewatering effluent amounts typically to 20 % of the incoming nitrogen load and therefore impacts significantly the energy demand for aeration. Biological pre-treatment of SDE in sidestream via nitrification/nitritation and returning the ammonia-reduced and nitrite/nitrate-rich effluent to the carbon extraction stage in main stream (1st stage of a 2-stage WWTP) can be a smart strategy to improve the energy balance of the WWTP. Nitrite and nitrate contain chemically bound oxygen which substitutes some of the oxygen demand for COD removal, thus reducing the energy demand for aeration. Additionally, in two-stage WWTPs the reduced carbon demand will increase the biogas yield in the anaerobic digestion.

The conversion of ammonia to nitrite or nitrate needs oxygen and aerobic conditions. The conversion of the formed nitrite or nitrate to elementary nitrogen needs raw wastewater, anoxic conditions and leads to a gain in oxygen, which reduces the demand of energy for aeration. The degradation path of ammonia to elementary nitrogen needs always the same demand of oxygen ($1,5\text{gO}_2/\text{gN}_2\text{-N}_{\text{removed}}$), irrespective of the bio-chemical process (nitrification/denitrification, nitritation/denitritation, anammox).

Detailed energy calculations show, that nitritation in SDE pre-treatment tank and the usage of chemically bound oxygen as an oxygen source in the 1st stage of the 2-stage WWTP appears to be the most advantageous and practicable option to save in energy demand for aeration (total energy for aeration: 9.6 kWh/PE/a). Since for further calculations of the energy demand of the WWTP Kirchbichl (demonstration plant in case study 5) operational data are required, their plausibility needs to be assured prior fitting the calibrated simulation model (dynamic modeling with SIMBA). Therefore, the WWTP was mass balanced for selected parameters (water flow, COD, TN and TP). Due to a lack of data for internal sludge streams an ex-extended sampling strategy was developed. The results of the total balances (gap <10 %) prior operational change (year 2015) and after changing the treatment of SDE from nitrification to nitritation (balanced period of 5 months) show the good quality of the data, obtained from the WWTP.

During the start-up phase of the nitritation with redirection of the SK 2 (return sludge from 2nd stage) from the SDE treatment tank to the 1st stage the addition of polymers (liquids and powders) was optimized. An overdosage of flocculants and the lack of dilution due to the bypassing of SK 2 probably cause the formation of foam during the aerated phase in the SDE treatment tank. Due to these problems, the start-up phase of the nitritation was investigated also at laboratory scale at 22 °C and 35 °C. A stable process of nitritation was observed after 15 days (35 °C) and 40 days (22 °C). The first representative samples from SDE treatment tank in Kirchbichl were taken 50 days after bypassing the SK 2 to the 1st stage. A stable nitritation without accumulation of nitrate was observed over the whole period. The conversion of $\text{NH}_4\text{-N}$ to $\text{NO}_x\text{-N}$ (mainly $\text{NO}_2\text{-N}$) under laboratory conditions as well as in full-scale ranged between 51.3 % and 52.9 %.



Due to the foam formation at the SDE treatment tank the N₂O measurements at WWTP Kirchbichl were postponed and conducted after the foam formation returned to normal. During this period a similar 2-stage HYBRID® WWTP, equipped with nitrification/denitrification tank for SDE treatment in sidestream, was monitored as reference. The N₂O measurements were performed in each biological treatment step of both WWTPs. The highest N₂O off-gas concentrations were found in the SDE treatment tanks, even if the maximal values at both WWTPs were quite different. In the 2nd stage of both WWTPs the N₂O off-gas concentrations showed a typical diurnal variation, related to the nitrogen load variation of the biological step. The lowest N₂O concentrations were detected in the 1st stage. Based on the N₂O concentrations and the aeration flow rates N₂O emission loads for each treatment tank will be estimated as the next step.



1. Introduction

1.1. Objectives of POWERSTEP and unique selling points

The project POWERSTEP is conceived to achieve a real paradigm shift in wastewater treatment processes: the overarching goal is to convert sewage treatment plants (STEPS) in power production facilities (POWER) while still achieving a high effluent quality of the treated wastewater.

Unique selling points of the POWERSTEP project:

1. **Breakthrough innovation:** the WWTP will be net energy producer! Wastewater as the last forgotten source of **biomass for renewable energy**.
2. No additional needs for **power infrastructure**, as WWTPs are already well connected in energy supply network and close to power demand (big cities).
3. First coordinated European project **demonstrating energy positive WWTPs** as cost effective combination of technological solutions.
4. Technical excellence and transdisciplinarity in consortium: pioneer concepts in cooperation between **wastewater and energy sectors**.
5. Demonstration with **first large-scale references**: Best practices for next generation WWTPs integrated with global assessment.
6. **Groundbreaking progress beyond State-of-the-Art**: > 50% increase in energy recovery in WWTPs documented by **'SMART' goals** (auditable, ambitioned and realistic) for each technology.
7. **Outstanding market and environment impact**: Global yearly market value of up \$30 Billion, energy cost savings for WWTP operators in Europe of at least €1.7 Billion per year and 5.9 Million tCO₂ reduction per year.
8. **Maximization of impact**: dedicated efforts on replication and dissemination targeting municipalities, regulatory bodies and international networks to accelerate **market penetration**

Municipal wastewater contains energy in form of organic carbon. The internal chemical energy content amounts to 14.4 MJ or 4 kWh per kg of chemical oxygen demand (COD). Based on the specific organic load of 120 g COD per population equivalent (PE) and year, the theoretical energy potential amounts to 175 kWh/PE/a. The electricity consumption to treat wastewater at large WWTPs is about 32 kWh per PE and year. Comparing both values, it becomes clear that municipal wastewater contains significantly more energy in its organic matter than required for its treatment.

A typical conventional WWTP consisting of primary sedimentation followed by an activated sludge process recovers around 10 % of this energy potential as electricity, mainly by anaerobic sludge digestion and subsequent biogas valorization in a combined Heat and Power plant (CHP).

An energy neutral or even an energy-positive wastewater treatment process is theoretically possible by converting this exploitable energy potential into useable energy forms, with the aim of cost efficiency for both, operators and public/industrial customers.

The high energy potential in municipal wastewater should be better exploited in future WWTP treatment schemes to save on limited fossil resources for electricity production and associated emissions of greenhouse gases, turning WWTPs into producers of renewable energy. The following Figure 1 shows the conventional, state-of-the-art wastewater treatment (figure above, power consumer) and a new concept for energy-positive wastewater treatment (figure below, power producer).



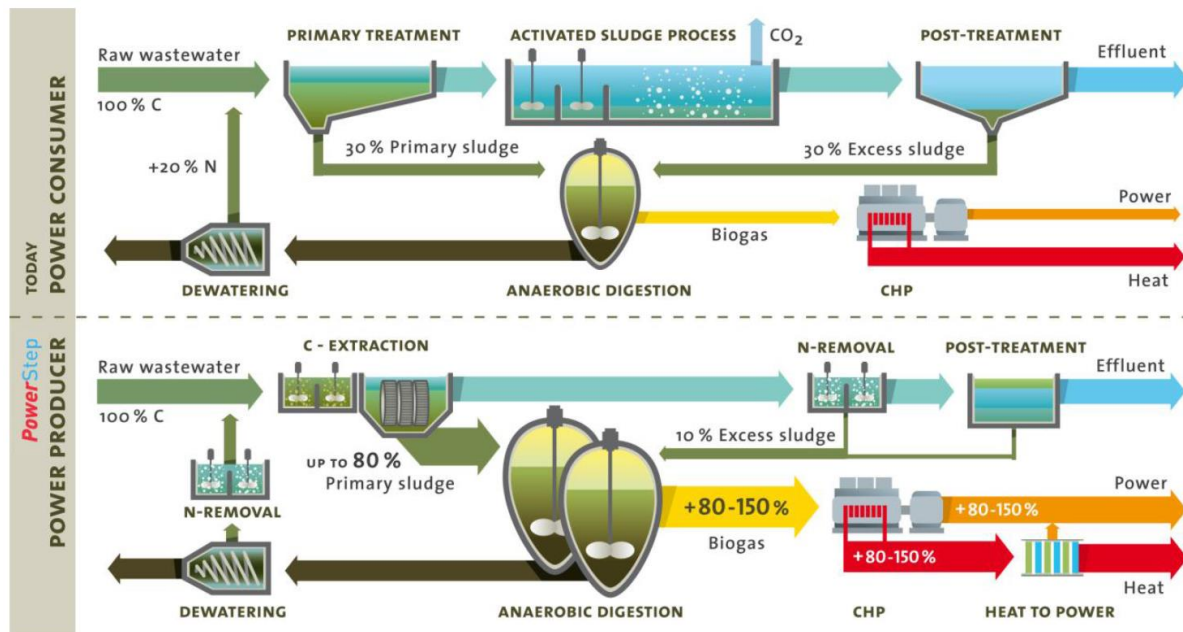


Figure 1: State-of-the-art of wastewater treatment and new concepts for energy-positive wastewater treatment

The concept to reach an energy-positive WWTP is based on the combination of innovative treatment concepts to remove organic carbon, nitrogen and phosphorous from the wastewater. Figure 2 shows the work packages included in the POWERSTEP project.

Work package 1 deals with organic carbon extraction for energy recovery. Technologies like upflow anaerobic sludge blanket reactors, advanced primary treatment or high load biosorption enhance carbon extraction with the specific goal of enhancing the conversion of organic carbon into biogas. Anaerobic digestion causes a release of nitrogen. Due to the higher amount of sludge/carbon in the digester, the amount of nitrogen increases, causing higher amount of ammonia in the occurring sludge dewatering effluent.

Work package 2 deals with the nitrogen removal in the main stream. Innovative technologies like main stream deammonification, or algae/duckweed treatment are tested. Conventional nitrification/denitrification can be also considered if the mass flow distribution of carbon between digestion and denitrification is optimized by advanced control systems.

Work package 3 deals with biogas valorisation and efficient energy management. The supplementary biogas can be either upgraded and compressed, or will generate more electricity in CHP plants, but also more excess heat, which in turn can be used for other complementary processes (thermal hydrolysis, ammonia stripping or distillation, sludge drying etc.). Excess heat can also be transferred into electricity through thermoelectric conversion.

Work package 4 deals with the biological or physical nitrogen treatment in sidestream. The nitrogen in the sludge dewatering effluent amounts typically to 20 % of the incoming nitrogen load and therefore impacts significantly the energy demand for aeration. Biological pre-treatment of SDE in sidestream via nitrification/nitritation and returning the ammonia-reduced and nitrite/nitrate-rich effluent to the carbon extraction stage in



main stream can be a smart strategy to improve the energy balance of the WWTP. Innovative approaches as the membrane ammonia stripping may not only remove but even recover ammonia as fertilizing product with minimized chemical demand and competitive investment costs.

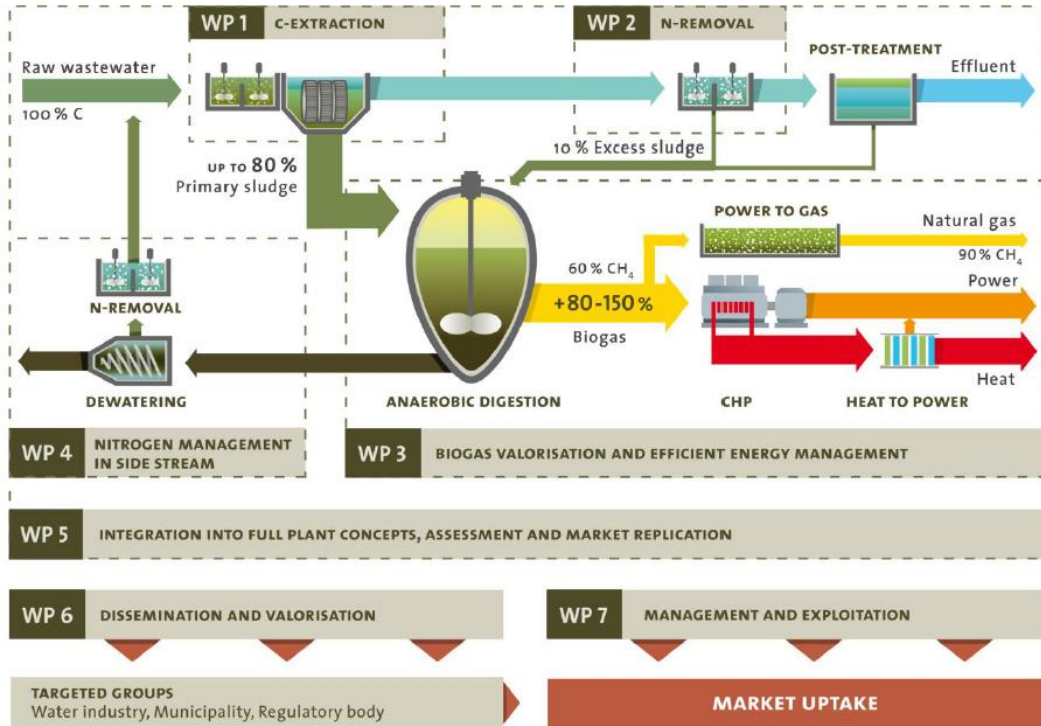


Figure 2: Concept of treatment scheme for energy-positive WWTP and the split into work packages

POWERSTEP will demonstrate the novel concepts and design treatment schemes of wastewater treatment plants that will be net energy producers, paving the way towards large implementation of such approaches and quick market penetration and supporting the business plans of participating technology providers.

Case study	Size of WWTP [pe]	WP1: Carbon extraction for energy recovery	WP2: Nitrogen removal in main stream	WP3: Biogas valorisation + energy management	WP4: Nitrogen management in side stream	WP5: Integration assessment and market replication	Site owner / operator / utility
1 Westewitz WWTP (GER)	2,000	X	X + X			X	OEWA Döbeln-Jahnatal
2 Källby WWTP (SE)	120,000	X	X			X	VASyd
3 Avedøre WWTP (DEN)	350,000			X		X	BIOFOS
4 Braunschweig WWTP (GER)	325,000			X + X	X	X	SEBS AVB
5 Kirchbichl WWTP (AT)	100,000	X			X	X	AWV Wörgl-Kirchbichl
6 Altenrhein WWTP (CH)	120,000				X	X	AVA Altenrhein

Figure 3: WWTPs for full-scale implementation of WP1-4



1.2. Description of content of WP 4

Work package 4 will explore new concepts for the treatment of ammonia in sludge dewatering effluent (SDE) which will save on aeration demand or even recover the nitrogen in form of a fertilizer product. In particular, two process configuration will be assessed at full-scale:

- **Nitrification** of SDE to NO_2 and recycle into the first stage of two-stage activated sludge plants (carbon extraction stage). Nitrification is a very stable and reliable process. Returning the NO_2 -rich effluent into the first high-rate activated sludge stage provides the required oxygen. Focusing on energy-sufficient two-stage activated sludge plants, strategies will be developed for the treatment and return of SDE depending on plant layout and sewage composition. Single-stage activated sludge plants will also be addressed, as well as a complete nitrification/denitrification system compared to a sidestream deammonification process.
- **Membrane ammonia stripping** with a novel membrane cross-flow system with low chemical demand, yielding a marketable nitrogen fertilizer product for nitrogen recovery from liquor.

SDE contains a major fraction, typically 15-20 % of the total nitrogen influent load in the form of ammonium (NH_4) and can be treated in main-stream or sidestream processes; i.e. directly in the main biological treatment step or in a separate treatment step. The general aim of the current sidestream process is a cost-efficient reduction of the N-load with reduced treatment tank volume, carbon demand and/or energy demand for aeration. In addition, nitrogen could also be recovered from SDE to produce a valuable fertilizer.

The conventional biological sidestream treatment options for SDE (Figure 4) reproduce long established wastewater treatment processes like nitrification/denitrification (over nitrate), or nitrification/denitrification (over nitrite). In these processes the organic carbon (i.e. COD) availability is essential for the transformation of nitrite or nitrate to N_2 gas. An alternative of the above processes is to pre-treat the SDE only by nitrification or nitrification prior to recycling the SDE to the main stream of the WWTP. Nitrite and nitrate contain chemically bound oxygen which substitutes some of the oxygen demand for COD removal, thus reducing the energy demand for aeration. Additionally, in two-stage WWTPs, the reduced carbon demand will increase the biogas yield in the anaerobic digestion.



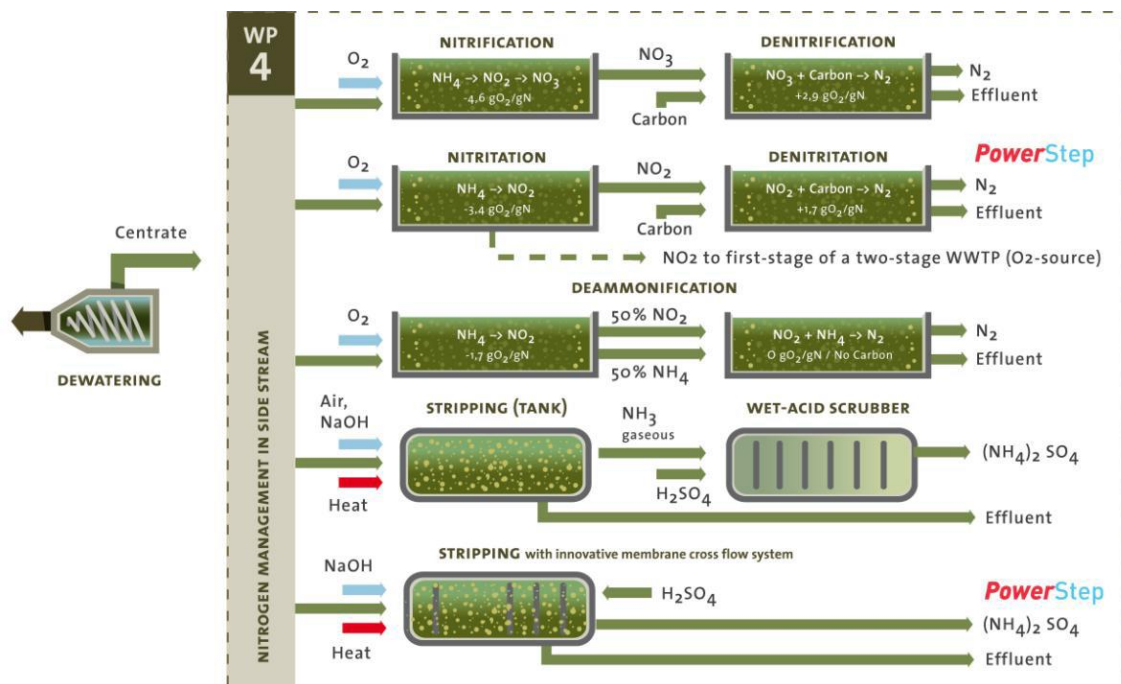


Figure 4: State-of-the-art approaches for sidestream NH_4 removal

Ammonia can also be physically removed from SDE by stripping ammonia from the water in gaseous form (NH_3), with typical efficiencies of $> 90\%$. In a two-step process, NH_3 is stripped by air from the SDE and subsequently removed from the gaseous phase in a wet-scrubber by adding sulfuric acid. An innovative method is to replace the spacious air loop system by a single cross-flow membrane that separates the SDE from the sulfuric acid, resulting in a compact treatment unit. Both methods yield a valuable nitrogen-sulfate fertilizer that can be spread directly on field without further processing or product certification. For an efficient stripping process, the equilibrium of NH_4/NH_3 needs to be tightly controlled by maintaining an elevated temperature (heat exchanger) and/or raising the pH (NaOH dosing). In POWERSTEP, the innovative membrane system for NH_3 stripping will be tested and evaluated in a large-scale unit (case study 6), thus moving beyond state-of-the-art in physical nitrogen treatment of SDE while recovering a valuable fertilizer.

The SMART goals of WP4:

- N return load: decrease total N load in WWTP influent by 20 %
- Nitritation of sludge water in two-stage plants: -10 % aeration in biological step, +5 % biogas production against dosing of untreated SDE into main stream
- Membrane stripping: -10 % in total costs compared to conventional air stripping

In case study 5 (Figure 5), POWERSTEP will integrate the stable process of nitritation in a two-stage WWTP in an optimised concept depending on WWTP layout and wastewater composition, reaching a significant decrease in energy required to treat the SDE nitrogen and improving carbon extraction and transfer into the digester.




Case Study 5: Kirchbichl WWTP (Austria)			
Size: 100,000 pe (15,000 m ³ /d) Site owner / operator: AWV Wörgl-Kirchbichl Case study leader: TUV			
			
Two-stage plant with anaerobic digestion in Kirchbichl (AT)			
Context	Medium-size two-stage WWTP (patented HYBRID process), strong seasonal variation due to tourism, with high nitrogen return loads in SDE. Integrated management of SDE targeted with stable and energy-efficient process to maximise energy recovery.		
Goals	<ul style="list-style-type: none"> - Full-scale demonstration of integrated SDE management via nitrification at a HYBRID two-stage WWTP, operational and long-term experience - Investigations on specific factors as e.g. C/N ratio or seasonal impacts - Process optimization by modelling and identification of technical/biological limits - Evaluation of the effects on the WWTP with regard to expected reduced aeration demand and increased biogas yield - Dynamic modelling of biological processes with electricity input/output - Comparison of nitrification with state-of-the-art processes in efficiency/costs 		
Prerequisite legal obligations	<ul style="list-style-type: none"> - Only Health and Safety approval 		
Work package	Technological solution	Description	Partners
WP4. Nitrogen management in side stream	Integrated Management of sludge process water for two-stage activated sludge plants	Nitrification of SDE and dosing into the first stage reduces aeration demand and increases biogas production compared to nitrification/denitrification	TUV

Figure 5: Case study 5 on WWTP Kirchbichl



2. Operating mode of a 2-stage WWTP

The biological step of a conventional 1-stage WWTP consists of an aeration tank and a clarifier (Figure 6). The nitrification process occurs under aerobic conditions in the biological step. In order to maximize the denitrification efficiency, the process is often carried out as pre-denitrification. For this purpose, nitrate is recirculated via an internal recirculation to the inflow area of the biological step, where raw wastewater with high COD content is discharged. Thickened sludge is returned from the clarifier to the anoxic zone of the aeration tank. In order to adjust the sludge age in the system a certain amount of biomass is removed as excess sludge.

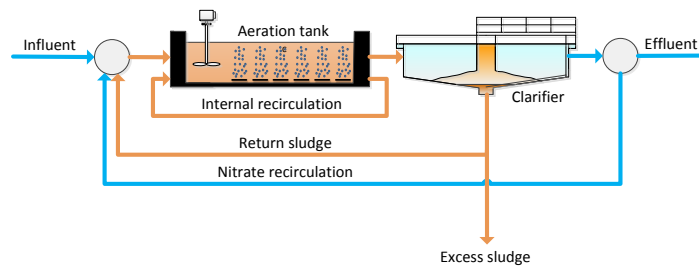


Figure 6: Flow scheme of a conventional 1-stage WWTP

WWTPs need to be upgraded usually due to lack of capacity and consequent insufficient removal efficiency. When on-site the space for the additional tanks is lacking, a 2-stage process configuration can be advantageous. (Figure 7). Each stage consists of aeration tanks and clarifiers, where the biomass settles and is retained and partially discharged, so that two completely separate sludge streams are enriched in the both stages. In the high-loaded 1st stage occurs the carbon removal, therefore a short sludge age is sufficient for the biological COD conversion. For the low-loaded 2nd stage, designed for nitrogen removal (nitrification/denitrification), a higher sludge age is required. Since organic carbon is partially removed in the 1st stage, a lack of carbon source in the 2nd stage can affect the denitrification. In this case a sidestream from the influent can be bypassed to the 2nd stage.

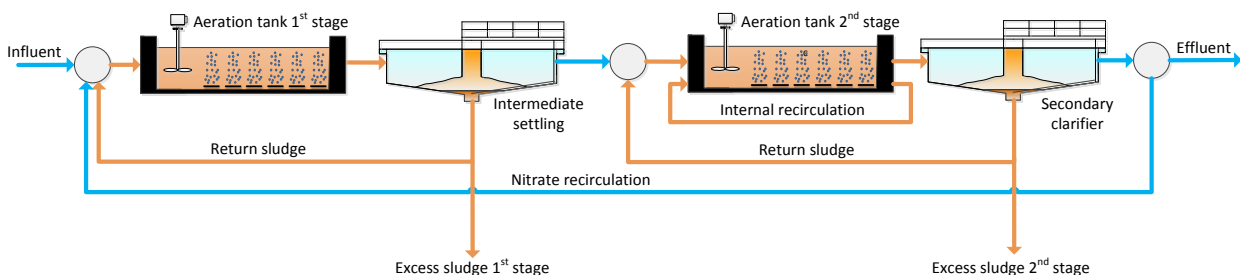


Figure 7: Flow scheme of a 2-stage WWTP

A modification of the 2-stage wastewater treatment process is the 2-stage HYBRID[®] process (Figure 8), which enables a high effluent quality at normal operational conditions as well as during shock loads, caused by industrial indirect discharger. The process consists of two activated sludge stages with different operational functions. The 1st stage is designed primarily for carbon removal, but also nitrification and denitrification occurs. The 2nd stage serves mainly the nitrogen removal process (nitrification and deni-



trification). The flexibility of the operation allows adaptation of the process to changing influent characteristics due to adjustment of the system for intensified carbon removal (1st stage) or nitrogen removal (2nd stage). This is possible due to the specific exchange of sludge between the both stages. With the partial sludge stream from the 1st stage to the 2nd stage organic carbon is provided and used there as substrate for the denitrification. The sludge stream from the 2nd stage in the 1st stage supplies biomass for sufficient nitrogen conversion in the 1st stage, which otherwise would not be possible due to the lower sludge age (carbon removal).

In general, this configuration combines the advantages of 1- and 2-stage wastewater treatment processes and is characterized by:

- Lower volume of the aeration tanks and lower space requirement respectively;
- Lower energy demand for aeration;
- Higher biogas yields of the digested sludge;
- Lower sludge index even at disadvantageous influent chemical composition (Wandl et al., 2002);
- Higher nitrogen removal efficiency (Wandl et al., 2002).

The both sludge circulation streams are negligible since they amount to only 3-5 % of the wastewater inflow and can be turned off in case of hydraulic peaks.

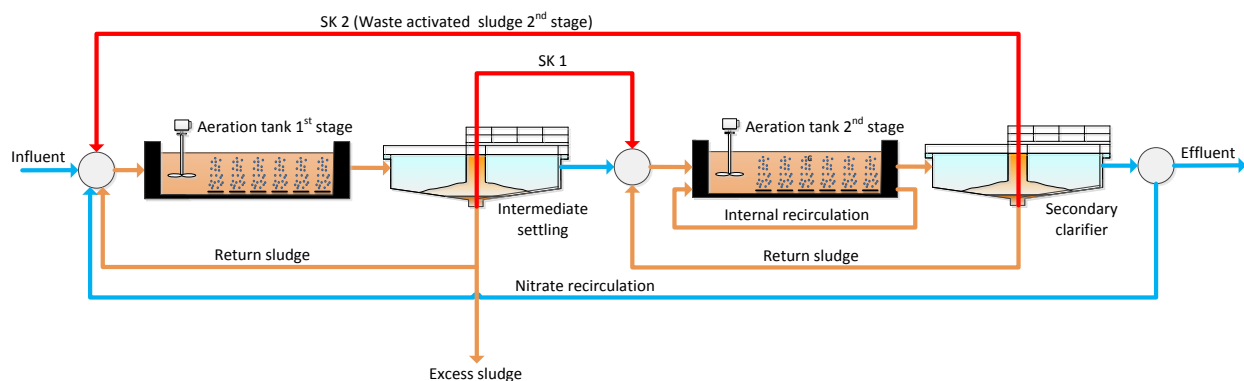


Figure 8: Flow scheme of a 2-stage HYBRID®-WWTP

The POWERSTEP-WWTP configuration (Figure 9) is very similar to that of the 2-stage WWTP. In the first step organic carbon is extracted from the influent. The organic carbon is biologically converted in a digester into biogas instead under aerobic conditions in the aeration tank, so that the COD content of the wastewater is used for heat and power production. To extract more COD via the primary sludge, a low sludge age is required, which corresponds to the high-loaded 1st stage of 2-stage WWTPs. The further treatment of the wastewater at the POWERSTEP-WWTP requires nitrogen removal in mainstream with reduced COD content of the wastewater (corresponds to the low-loaded 2nd stage of 2-stage WWTPs).



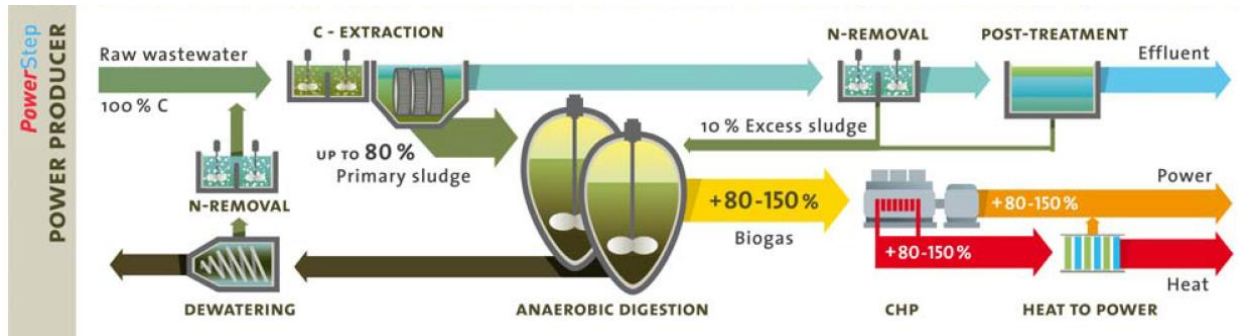


Figure 9: Flow scheme of the POWERSTEP WWTP

Since on WWTPs with sludge digestion the SDE is returned back to the influent without any treatment, but this operational step is considered in the POWERSTEP concept, the sludge management was implemented in the flow scheme of a 2-stage WWTP (Figure 10). Regarding the goals of WP 4 and CS 5 the nitrogen removal in sidestream is depicted as nitrification process.

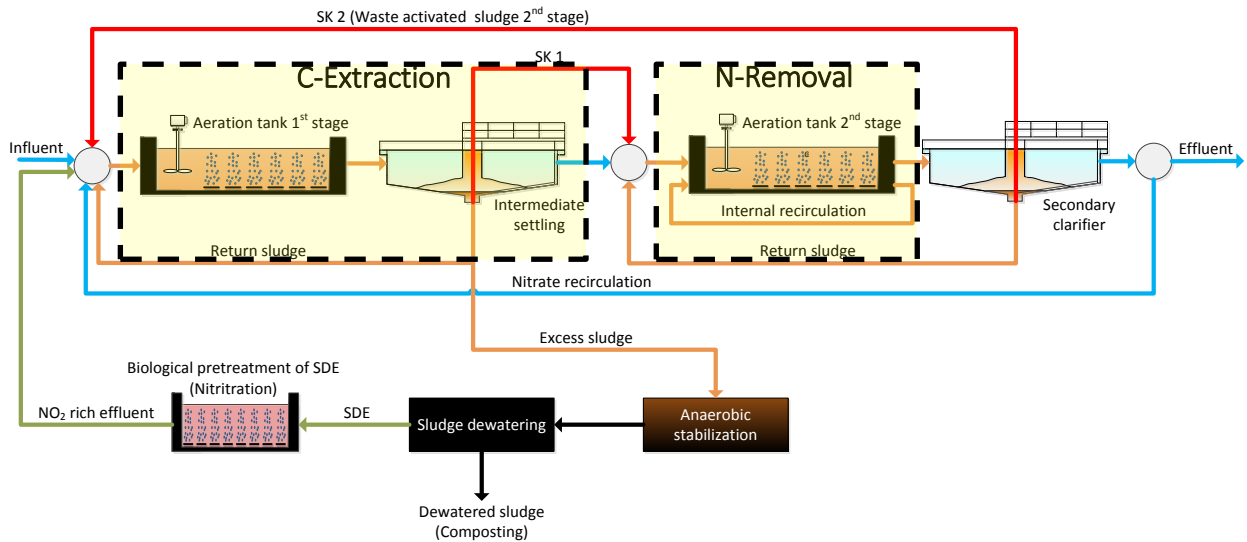


Figure 10: Flow scheme of 2-stage WWTP with nitrogen removal in sidestream

The 2-stage WWTP Kirchbichl (CS 5) fulfils the specific requirements of the POWERSTEP-WWTP regarding the plant configuration. Additionally, the WWTP is situated in a winter tourism region so the wastewater treatment performance of the modified operation (POWERSTEP concept) can be proved under unfavorable conditions (low temperature, high ammonia load, low C/N ratio).



3. Treatment of sludge dewatering effluent

Sludge dewatering effluent (SDE) is a term coined to describe the water resulting from the process of de-watering an anaerobically digested sludge. SDE contains a major fraction, typically 15-20 %, of the total nitrogen influent load in the form of ammonium (NH_4). Several biological, physical or chemical procedures have been engineered to treat this significant additional load of ammonium-nitrogen. SDE can be treated in mainstream or sidestream processes, i.e. directly in the main biological treatment step or in a separate treatment step before recycling it to the main stream. The general aim of the current sidestream processes is a cost-efficient reduction of the N-load. This can be done by reducing reactor volume, carbon demand, energy demand for aeration, or a combination of these. In addition to that, nitrogen could also be recovered to produce a marketable fertilizer.

3.1. Physical treatment of SDE

A physical SDE treatment process is to strip the ammonia from the water; this process has a typical efficiency of up to 90 %. This can be done by a two-step process, where ammonia (NH_3) is stripped by air from the SDE and subsequently removed from the gaseous phase in a wet-scrubber by adding sulfuric acid (Lei et al., 2007; Guštin and Marinšek-Logar, 2011). An innovative method is to replace the voluminous air loop system by a single cross-flow membrane that separates the SDE from the sulfuric acid, resulting in a compact treatment unit. Both methods yield a valuable nitrogen-sulfate fertilizer. For an efficient stripping process, the equilibrium of ammonium (NH_4) and ammonia (NH_3) needs to be tightly controlled by maintaining an appropriate (usually elevated) temperature and/or raising the pH of the SDE. Therefore, the dosing of chemicals (NaOH) or a heat-exchange-unit may be needed. The development and full-scale application of this innovative stripping method will be part of this work package (WP4).

3.2. Chemical treatment of SDE

A chemical SDE treatment process is to precipitate the nitrogen in the form of a low soluble salt like Magnesium-Ammonium-Phosphate (MAP, MgNH_4PO_4). The necessary phosphate (PO_4) and alkaline earth metals are usually present in the SDE, however only to a low extent, resulting in a low yield of ~5 % (Doyle and Parsons, 2002; Demirer and Othman, 2009.). To increase the ammonium removal rate, a phosphate source (e.g. phosphoric acid) and magnesium salts have to be additionally dosed as precipitants.



3.3. Biological treatment of SDE

At the first step elimination of ammonium in SDE assumes the oxidation of NH_4 to nitrite or nitrate, irrespective of what plant configuration (conventional WWTP or two-stage WWTP) is used. This conversion to NO_2 and/or NO_3 can take place in the mainstream of the WWTP or in a separate sidestream treatment tank, which implies higher nitrogen content. Dosing the SDE to the mainstream causes both, higher demand of oxygen for conversion of NH_4 and a higher demand of COD for nitrogen removal in the biological step. This may lead to several problems. The following Figure 11 shows a conventional WWTP with treatment of the SDE in the mainstream.

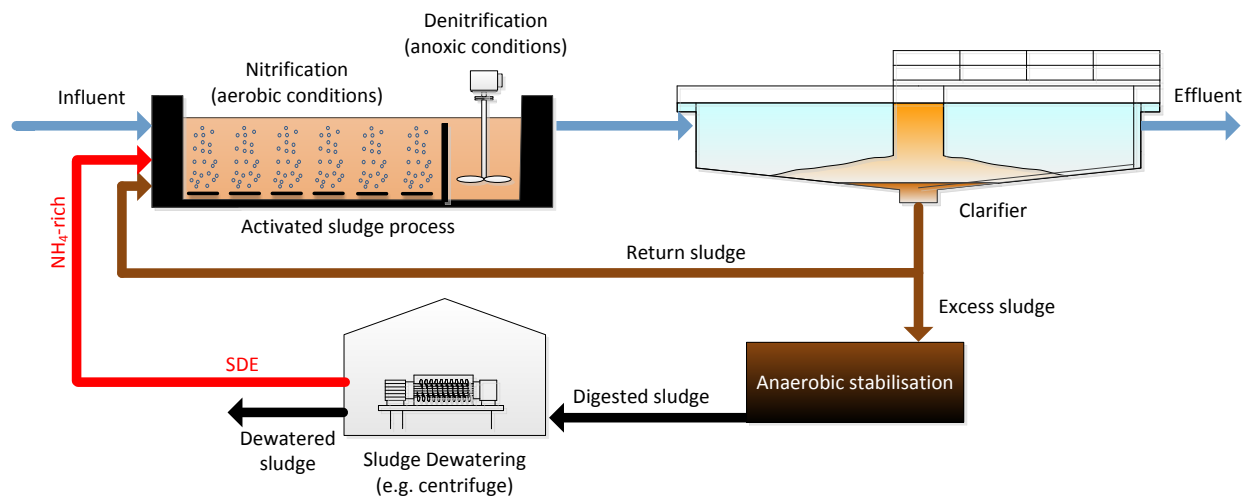


Figure 11: Treatment of SDE in mainstream

The unit for sludge dewatering is usually in use during the day. The SDE is directly returned to the biological treatment step, if there is no storage possibility. Thus, due to the increased ammonia load, the demand of aeration in the biological step increases in order to meet the threshold value in the effluent. Due to the increasing aerated zone, the retention time of the biomass under aerobic conditions rises. This enables the conversion of a higher amount of ammonia to nitrate. Furthermore, the expansion of the aerated zone leads to a higher COD reduction in the biological step. As a result, the amount of readily biodegradable carbon for denitrification decreases and the energy content of the excess sludge is reduced. This results in a lower biogas yield in the digester. Additionally, due to the recirculated high nitrogen load from dewatering and the additional required capacity for nitrogen oxidation in the biological step, the anoxic volume for denitrification decreases, causing deterioration of nitrogen removal.

Due to the pre-treatment of the SDE, the performance of the WWTP can be significantly improved. The following Figure 12 shows the flow scheme of a WWTP with pre-treatment of SDE.



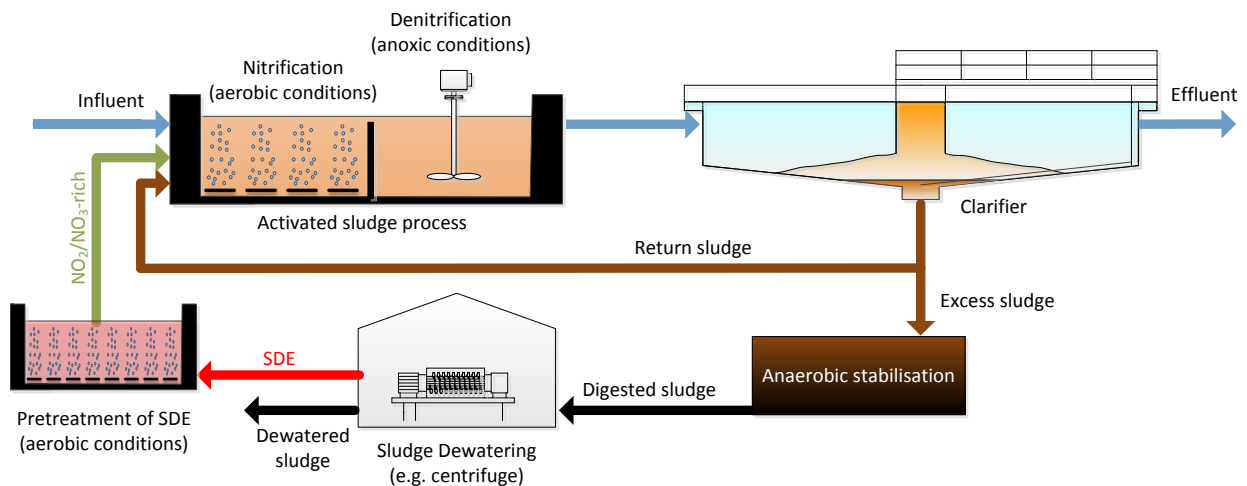


Figure 12: Pre-treatment of SDE in sidestream and dosing to mainstream

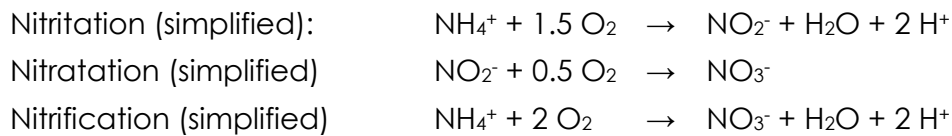
Compared to Figure 11, the flow scheme of the WWTP in Figure 12 is extended by an additional tank for pre-treatment of the effluent from sludge dewatering. The pre-treatment of the ammonia-rich SDE in sidestream leads to a lower ammonia load to the biological step. The lower amount of ammonia reduces the aerated zone and ensures a sufficient denitrification due to enlarged anoxic capacity. In addition, the amount of readily biodegradable carbon for denitrification increases due to the reduction of the aerobic sludge age.

Biological sidestream treatment options for high ammonium loaded process water are either methods and biocenosis conventionally applied in wastewater treatment as e.g. nitrification/denitrification (Gustavsson, 2011) and nitrification/denitrification (Fux et al., 2006; Abeling, 1994) or more specific processes and bacteria as by the de-ammonification process (also known as Anammox) (Hippen, 2001; Twachtmann, 1999), (Daigger et al., 2011; Magrí et al., 2013; Mouilleron et al., 2014; Lackner et al., 2014) (Demon)).

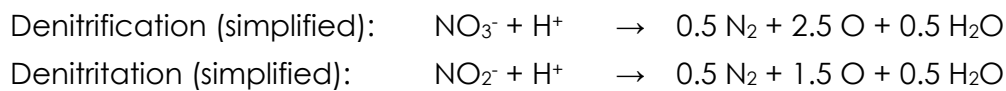
Nitrification is a process by which ammonium/ammonia ($\text{NH}_4^+/\text{NH}_3$) is oxidized into nitrite (NO_2^-) and nitrate (NO_3^-). The first step of the conversion from ammonium (or ammonia) to nitrite is done by ammonia-oxidizing bacteria (AOB, Nitrosomonas). The nitrite-oxidizing bacteria (NOB, Nitrobacter) are responsible for the further oxidation from nitrite to nitrate. The two processes are called nitritation ($\text{NH}_4^+ \rightarrow \text{NO}_2^-$) and nitratation ($\text{NO}_2^- \rightarrow \text{NO}_3^-$). These bacterial groups are autotrophic aerobic organisms, which means that both, dissolved oxygen and carbon dioxide (CO_2) (or in practice bicarbonate, HCO_3^-) as a carbon source is essential for the processes of nitritation and nitratation. Both process steps run usually simultaneously but can also run at different rates. If the process of nitritation is faster than the nitratation, the nitrite concentration and the corresponding undissociated nitrous acid (HNO_2) will increase. Nitrous acid is beside ammonia the main factor for inhibition of AOB and NOB. The group of NOB (conversion of NO_2^- to NO_3^-) seems to be more sensitive to ammonia inhibition than the bacteria group of AOB (Buday et al., 1999). This inhibition of the NOB leads to occurrence of nitrite, if ammonia concentration increases.



The following chemical equations describes the partial steps of nitrification (nitritation and nitrataion) and the process as a whole:

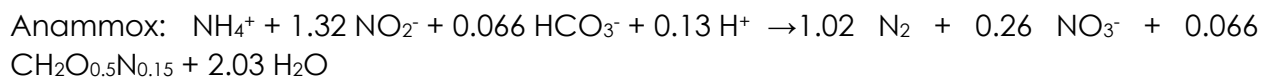


The formed nitrite or nitrate is subsequent converted to nitrogen via reductase by certain heterotrophic and autotrophic bacteria using different organic compounds in the raw wastewater as an electron donor.



The processes of denitrification and denitrataion need both easily degradable organic compounds in appropriate quantity and anoxic conditions.

Beside these conventional processes a new species of bacteria (anammox) was detected in the 1980ies. The anammox-reaction (anaerobic ammonia oxidation) is a two-step process. In the first step ammonia is oxidized under aerobic conditions to nitrite. In the second step ammonia and nitrite are converted directly into nitrogen and water without the usage of organic carbon. The following equation shows the process of anammox (Strous et al., 1998):



The first step of nitrification, nitritation and deammonification is the conversion of ammonium (NH_4^+) under aerobic conditions to nitrite (NO_2^-). So far, energy consumption for aeration is equal. For further oxidation from nitrite to nitrate (NO_3^- ; nitrification) additional energy consumption for aeration is necessary.

The conversion of 1 mol NH_4^+ to 1 mol NO_3^- needs 2 mol O_2 . With regard to this, the oxygen demand for nitrification of 1 g $\text{NH}_4\text{-N}$ to 1g $\text{NO}_3\text{-N}$ is calculated to 4.57 g O_2 . This stoichiometric oxygen demand is reduced in practice due to the usage of carbon dioxide dissolved in water as an electron donor (0.24 g $\text{O}_2/\text{g NO}_3\text{-N}$). This leads to an effective oxygen demand for nitrification of 4.33 g $\text{O}_2/\text{g NO}_3\text{-N}$.

The process of nitritation requires 1.5 mol O_2 for the conversion of 1 mol NH_4^+ to 1 mol NO_2^- . This leads to an oxygen consumption of 3.43 g O_2 for the conversion of 1 g $\text{NH}_4\text{-N}$ to 1 g $\text{NO}_2\text{-N}$. The bacterial usage of dissolved CO_2 reduces in practice the oxygen demand to 3.25 g $\text{O}_2/\text{g NO}_2\text{-N}$ ($0.24 \cdot 3/4 = 0.18$ g $\text{O}_2/\text{g NO}_2\text{-N}$).

The reduction of the chemically bound nitrogen in 1 mol NO_3^- to elementary nitrogen (denitrification) leads to a gain of oxygen in the amount of 2.5 mol O_2 . According to that, the oxygen yield by denitrification of 1 g $\text{NO}_3\text{-N}$ can be calculated to 2.86 g O_2 . If nitrite is the electron acceptor, the yield in oxygen due to denitrataion can be calculated to 1.71 g $\text{O}_2/1$ g $\text{NO}_2\text{-N}$.



The conversion of ammonia to nitrite or nitrate needs oxygen and aerobic conditions. The conversion of the formed nitrite or nitrate to elementary nitrogen needs raw wastewater, anoxic conditions and leads to a gain in oxygen, which reduces the demand of energy for aeration. The degradation path of ammonia to elementary nitrogen needs always the same demand of oxygen, irrespective of the biochemical process. To show the equality of the different pathways the following figures (Figure 13, Figure 14, Figure 15) are created. It is shown, that the oxygen consumption for the different treatment options (nitrification/denitrification, nitritation/denitritation and anammox) is the same.

The sludge dewatering effluent is pre-treated in a sidestream tank under permanent aerobic conditions. The described treatment options are nitrification in sidestream, nitritation in side stream and nitritation/anammox in sidestream. First results of SDE treatment in laboratory scale (5.2.2) have shown that a conversion from ammonia to nitrite or nitrate is limited by the alkalinity and only 55 % of the incoming $\text{NH}_4\text{-N}$ which means, that 55 % of the incoming $\text{NH}_4\text{-N}$ can be converted to $\text{NO}_3\text{-N}$ or $\text{NO}_2\text{-N}$. The effluent of the sidestream treatment is recirculated to the mainstream, where further nitrification of the remaining $\text{NH}_4\text{-N}$ and denitrification or denitritation of the formed $\text{NO}_3\text{-N}$ and $\text{NO}_2\text{-N}$ is ensured.

The following table (Table 1) summarizes the results from the calculation shown in Figure 13, Figure 14 and Figure 15.

Table 1: Oxygen consumption and oxygen yield for the conversion of 1 g $\text{NH}_4\text{-N}$ via different treatment options in side stream and dosing the effluent in the main stream

SDE pre-treatment option	O_2 -consumption	O_2 -yield	total
Nitrification/Denitrification	4.33 g O_2	2.86 g O_2	1.5 g $\text{O}_2/\text{g N}_2\text{-N}_{\text{removed}}$
Nitritation/Denitritation	3.74 g O_2	2.23 g O_2	1.5 g $\text{O}_2/\text{g N}_2\text{-N}_{\text{removed}}$
Anammox	1.93 g O_2	0.40 g O_2	1.5 g $\text{O}_2/\text{g N}_2\text{-N}_{\text{removed}}$



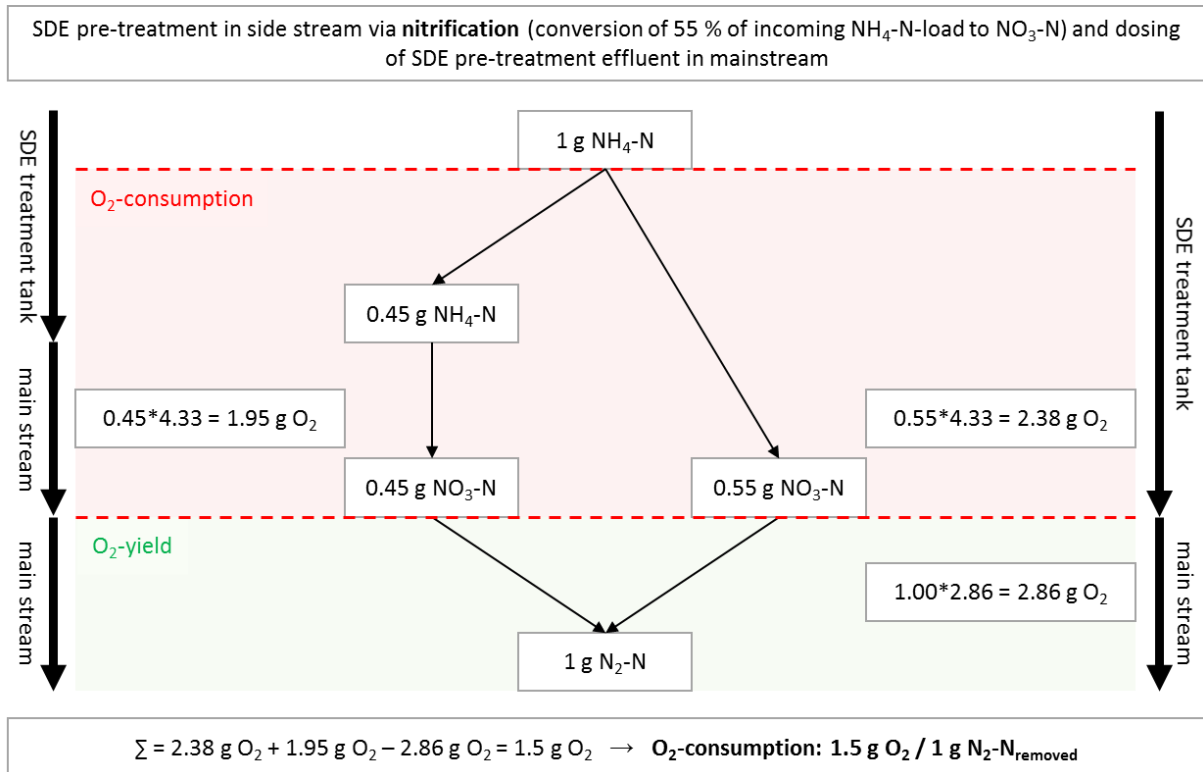


Figure 13: Treatment of SDE via nitrification in sidestream and recirculation to mainstream

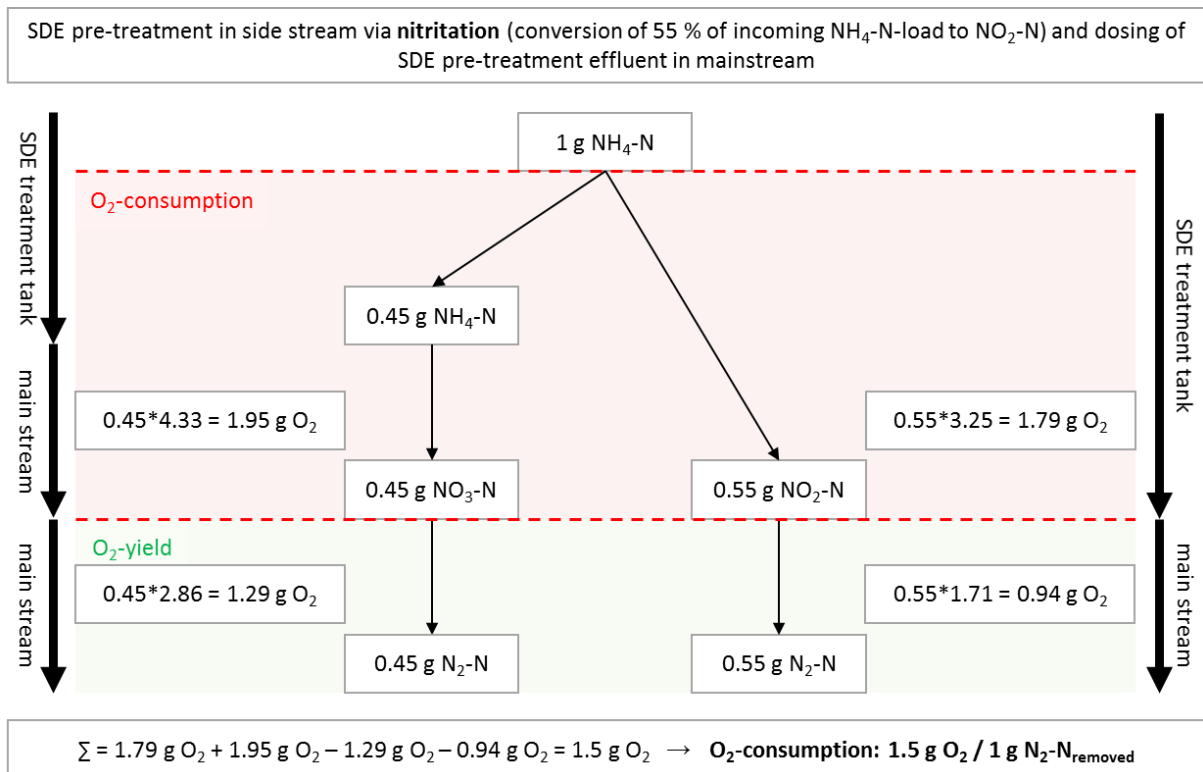


Figure 14: Treatment of SDE via nitritation in sidestream and recirculation to mainstream



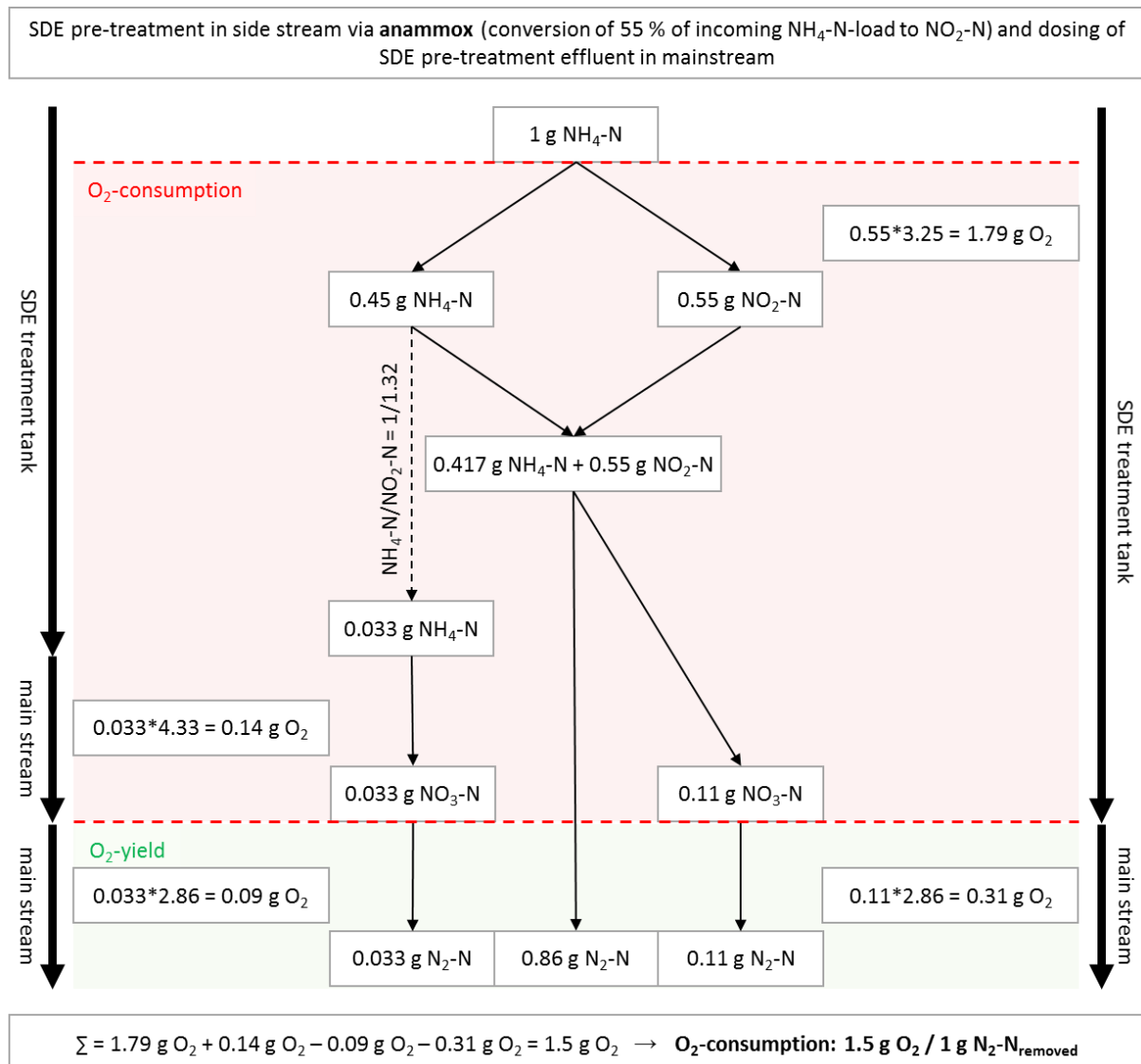


Figure 15: Treatment of SDE via anammox in sidestream and recirculation to mainstream

The oxygen demand for nitrogen removal is equal and irrespective of the biochemical pathway. The energy demand for nitrogen removal depends on the WWTP configuration. The following figures (Figure 16, Figure 17, Figure 18, Figure 19 and Figure 20) show different configurations of WWTP (1-stage, 2-stage) with different kinds of SDE treatment (without pre-treatment, pre-treatment in sidestream via nitrification, nitrification and anammox) and the related COD balances. Based on this, the energy from biogas utilisation and the overall energy for aeration is calculated. These calculations of energy demand and energy gain show, that the chosen treatment option for SDE is crucial for the energy efficiency of the whole wastewater treatment processes, although the oxygen demand for nitrogen removal is the same.



Table 2: Assumptions for calculation of 1-stage WWTP without SDE pre-treatment

Influent COD	120 g/PE/d	C _s (oxygen saturation)	10 mg/L
Dewatered sewage sludge COD	30 g/PE/d	C _x (oxygen concentration)	1.5 mg/L
Effluent COD	7.5 %	Oxygen transfer efficiency in clean water	3.6 kgO ₂ /kWh
Influent N	8 g/PE/d	a-factor 1 st stage	0 -
Dewatered sewage sludge N	2 g/PE/d	a-factor 2 nd stage	0.6 -
Ratio N/COD _{digested}	5 %	a-factor SDE treatment tank	0 -
Efficiency of N-removal	80 %	COD respiration in the 1 st stage	0 %
Efficiency of preliminary clarification	30 %	COD respiration in the 2 nd stage	60 %
COD removal 1 st stage	0 %	Energy from biogas	3 kWh/m ³ CH ₄

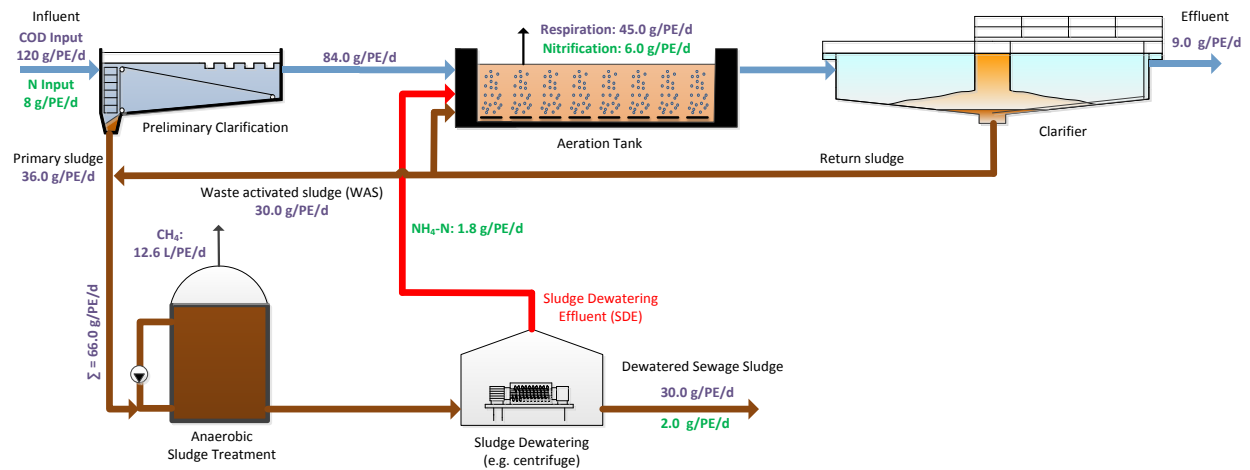


Figure 16: COD balance of a 1-stage WWTP without SDE pre-treatment

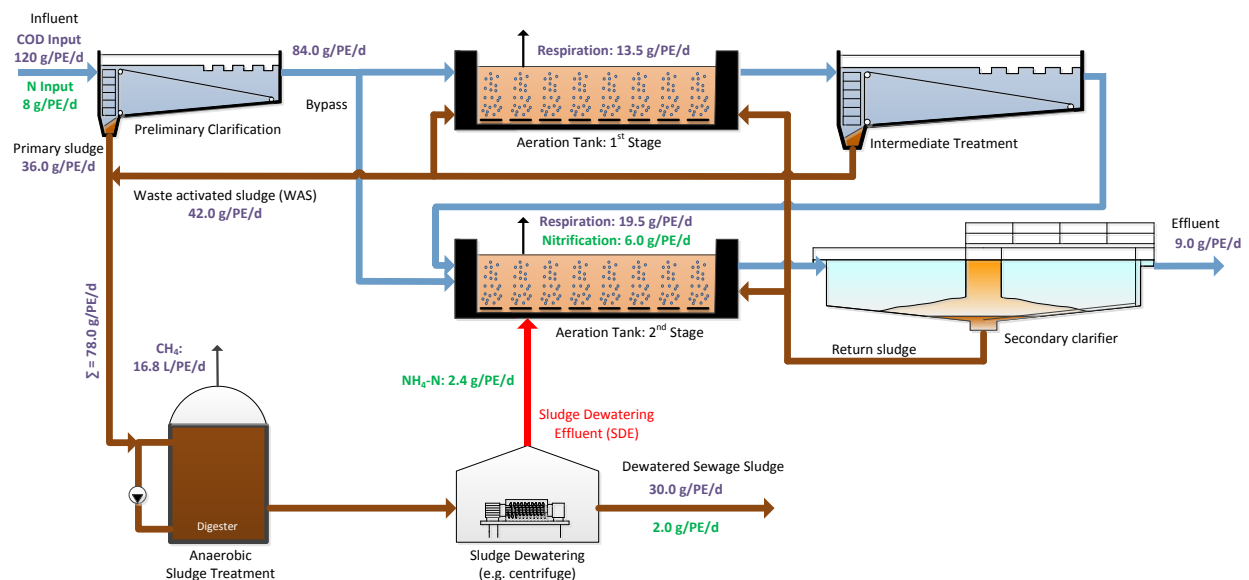
Table 3: Results of the calculation of energy consumption for aeration and energy from biogas utilisation

Respired COD in the 1 st stage	0 g/PE/d	Energy for aeration in the 1 st stage	0 Wh/PE/d
Respired COD in the 2 nd stage	45.0 g/PE/d	Energy for aeration in the 2 nd stage	31.8 Wh/PE/d
CH ₄ in biogas	12.6 L/PE/d	Energy for aeration in the SDE treatment tank	0 Wh/PE/d
O ₂ -demand for nitrification 2 nd stage	26.0 g/PE/d	Energy for aeration overall	31.8 Wh/PE/d
O ₂ -demand for SDE treatment tank	0 g/PE/d	Energy for aeration overall	11.6 kWh/PE/a
O ₂ -savings in the 1 st stage	0 g/PE/d	Energy from biogas utilisation	37.8 Wh/PE/d
O ₂ -savings in the 2 nd stage	12.6 g/PE/d	Energy from biogas utilisation	13.8 kWh/PE/a



Table 4: Assumptions for calculation of 2-stage WWTP without SDE pre-treatment

Influent COD	120 g/PE/d	C_s (oxygen saturation)	10 mg/L
Dewatered sewage sludge COD	30 g/PE/d	C_x (oxygen concentration)	1.5 mg/L
Effluent COD	7.5 %	Oxygen transfer efficiency in clean water	3.6 kgO ₂ /kWh
Influent N	8 g/PE/d	a-factor 1 st stage	0.4 -
Dewatered sewage sludge N	2 g/PE/d	a-factor 2 nd stage	0.6 -
Ratio N/COD _{digested}	5 %	a-factor SDE treatment tank	0 -
Efficiency of N-removal	80 %	COD respiration in the 1 st stage	30 %
Efficiency of preliminary clarification	30 %	COD respiration in the 2 nd stage	65 %
COD removal 1 st stage	60 %	Energy from biogas	3 kWh/m ³ CH ₄


Figure 17: COD balance of a 2-stage WWTP without SDE pre-treatment
Table 5: Results of the calculation of energy consumption for aeration and energy from biogas utilisation

Respired COD in the 1 st stage	13.5 g/PE/d	Energy for aeration in the 1 st stage	11.0 Wh/PE/d
Respired COD in the 2 nd stage	19.5 g/PE/d	Energy for aeration in the 2 nd stage	17.9 Wh/PE/d
CH ₄ in biogas	16.8 L/PE/d	Energy for aeration in the SDE treatment tank	0 Wh/PE/d
O ₂ -demand for nitrification 2 nd stage	26.0 g/PE/d	Energy for aeration overall	28.9 Wh/PE/d
O ₂ -demand for SDE treatment tank	0 g/PE/d	Energy for aeration overall	10.6 kWh/PE/a
O ₂ -savings in the 1 st stage	0 g/PE/d	Energy from biogas utilisation	50.4 Wh/PE/d
O ₂ -savings in the 2 nd stage	12.6 g/PE/d	Energy from biogas utilisation	18.4 kWh/PE/a



Table 6: Assumptions for calculation of 2-stage WWTP with SDE pre-treatment in sidestream via nitrification

Influent COD	120 g/PE/d	C _s (oxygen saturation)	10 mg/L
Dewatered sewage sludge COD	30 g/PE/d	C _x (oxygen concentration)	1.5 mg/L
Effluent COD	7.5 %	Oxygen transfer efficiency in clean water	3.6 kgO ₂ /kWh
Influent N	8 g/PE/d	a-factor 1 st stage	0.4 -
Dewatered sewage sludge N	2 g/PE/d	a-factor 2 nd stage	0.6 -
Ratio N/COD _{digested}	5 %	a-factor SDE treatment tank	0.7 -
Efficiency of N-removal	80 %	COD respiration in the 1 st stage	25 %
Efficiency of preliminary clarification	30 %	COD respiration in the 2 nd stage	65 %
COD removal 1 st stage	60 %	Energy from biogas	3 kWh/m ³ CH ₄

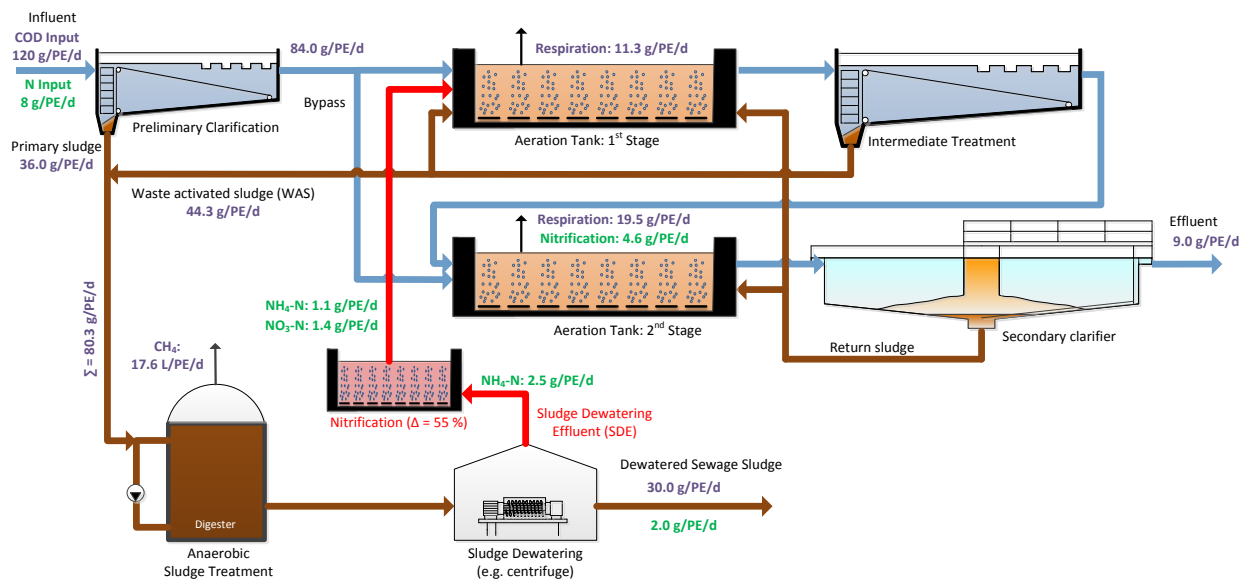


Figure 18: COD balance of a 2-stage WWTP with SDE pre-treatment in sidestream via nitrification

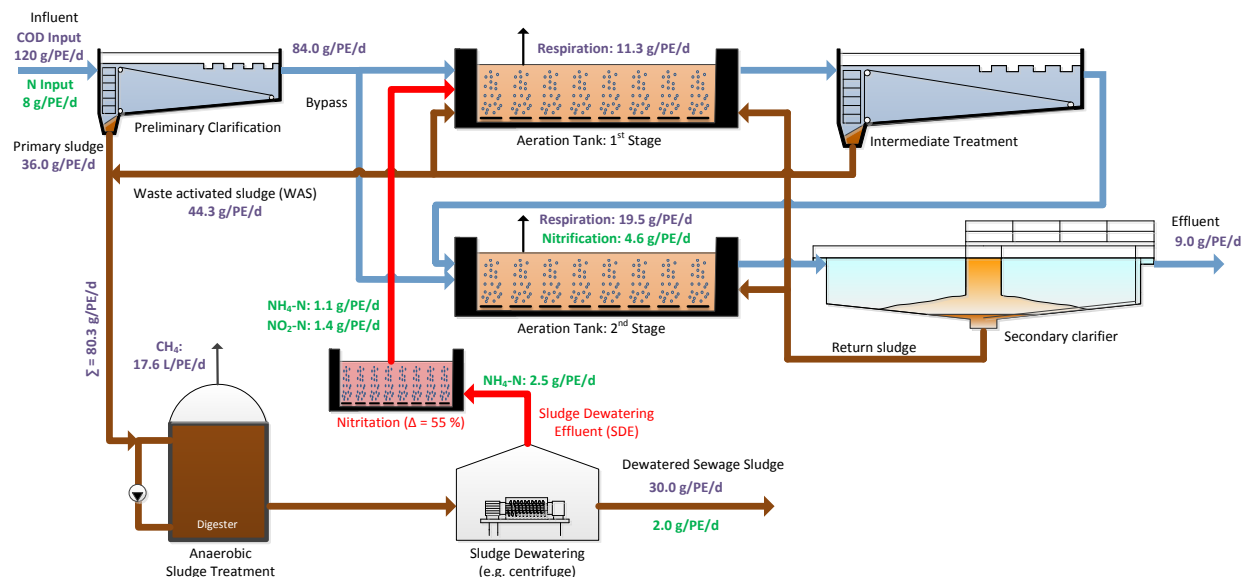
Table 7: Results of the calculation of energy consumption for aeration and energy from biogas utilisation

Respired COD in the 1 st stage	11.3 g/PE/d	Energy for aeration in the 1 st stage	6.0 Wh/PE/d
Respired COD in the 2 nd stage	19.5 g/PE/d	Energy for aeration in the 2 nd stage	16.8 Wh/PE/d
CH ₄ in biogas	17.6 L/PE/d	Energy for aeration in the SDE treatment tank	2.8 Wh/PE/d
O ₂ -demand for nitrification 2 nd stage	20.0 g/PE/d	Energy for aeration overall	25.6 Wh/PE/d
O ₂ -demand for SDE treatment tank	6.0 g/PE/d	Energy for aeration overall	9.3 kWh/PE/a
O ₂ -savings in the 1 st stage	4.0 g/PE/d	Energy from biogas utilisation	52.8 Wh/PE/d
O ₂ -savings in the 2 nd stage	8.6 g/PE/d	Energy from biogas utilisation	19.3 kWh/PE/a



Table 8: Assumptions for calculation of 2-stage WWTP with SDE pre-treatment in sidestream via nitrification

Influent COD	120 g/PE/d	C _s (oxygen saturation)	10 mg/L
Dewatered sewage sludge COD	30 g/PE/d	C _x (oxygen concentration)	1.5 mg/L
Effluent COD	7.5 %	Oxygen transfer efficiency in clean water	3.6 kgO ₂ /kWh
Influent N	8 g/PE/d	a-factor 1 st stage	0.4 -
Dewatered sewage sludge N	2 g/PE/d	a-factor 2 nd stage	0.6 -
Ratio N/COD _{digested}	5 %	a-factor SDE treatment tank	0.7 -
Efficiency of N-removal	80 %	COD respiration in the 1 st stage	25 %
Efficiency of preliminary clarification	30 %	COD respiration in the 2 nd stage	65 %
COD removal 1 st stage	60 %	Energy from biogas	3 kWh/m ³ CH ₄


Figure 19: COD balance of a 2-stage WWTP with SDE pre-treatment in sidestream via nitrification
Table 9: Results of the calculation of energy consumption for aeration and energy from biogas utilisation

Respired COD in the 1 st stage	11.3 g/PE/d	Energy for aeration in the 1 st stage	7.3 Wh/PE/d
Respired COD in the 2 nd stage	19.5 g/PE/d	Energy for aeration in the 2 nd stage	16.8 Wh/PE/d
CH ₄ in biogas	17.6 L/PE/d	Energy for aeration in the SDE treatment tank	2.1 Wh/PE/d
O ₂ -demand for nitrification 2 nd stage	20.0 g/PE/d	Energy for aeration overall	26.2 Wh/PE/d
O ₂ -demand for SDE treatment tank	4.5 g/PE/d	Energy for aeration overall	9.6 kWh/PE/a
O ₂ -savings in the 1 st stage	2.4 g/PE/d	Energy from biogas utilisation	52.8 Wh/PE/d
O ₂ -savings in the 2 nd stage	8.6 g/PE/d	Energy from biogas utilisation	19.3 kWh/PE/a



Table 10: Assumptions for calculation of 2-stage WWTP with SDE pre-treatment in sidestream via deammonification

Influent COD	120 g/PE/d	C _s (oxygen saturation)	10 mg/L
Dewatered sewage sludge COD	30 g/PE/d	C _x (oxygen concentration)	1.5 mg/L
Effluent COD	7.5 %	Oxygen transfer efficiency in clean water	3.6 kgO ₂ /kWh
Influent N	8 g/PE/d	a-factor 1 st stage	0.4 -
Dewatered sewage sludge N	2 g/PE/d	a-factor 2 nd stage	0.6 -
Ratio N/COD _{digested}	5 %	a-factor SDE treatment tank	0.7 -
Efficiency of N-removal	80 %	COD respiration in the 1 st stage	25 %
Efficiency of preliminary clarification	30 %	COD respiration in the 2 nd stage	65 %
COD removal 1 st stage	60 %	Energy from biogas	3 kWh/m ³ CH ₄

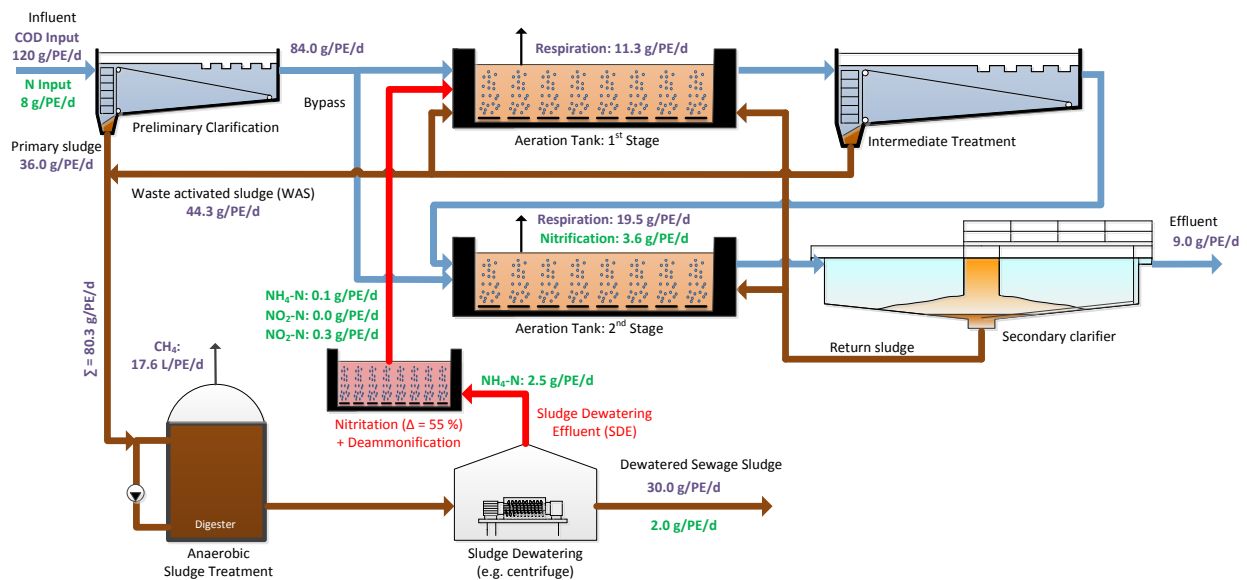


Figure 20: COD balance of a 2-stage WWTP with SDE pre-treatment in sidestream via deammonification

Table 11: Results of the calculation of energy consumption for aeration and energy from biogas utilisation

Respired COD in the 1 st stage	11.3 g/PE/d	Energy for aeration in the 1 st stage	8.6 Wh/PE/d
Respired COD in the 2 nd stage	19.5 g/PE/d	Energy for aeration in the 2 nd stage	16.0 Wh/PE/d
CH ₄ in biogas	17.6 L/PE/d	Energy for aeration in the SDE treatment tank	2.1 Wh/PE/d
O ₂ -demand for nitrification 2 nd stage	15.5 g/PE/d	Energy for aeration overall	26.6 Wh/PE/d
O ₂ -demand for SDE treatment tank	4.5 g/PE/d	Energy for aeration overall	9.7 kWh/PE/a
O ₂ -savings in the 1 st stage	0.8 g/PE/d	Energy from biogas utilisation	52.8 Wh/PE/d
O ₂ -savings in the 2 nd stage	5.6 g/PE/d	Energy from biogas utilisation	19.3 kWh/PE/a



The following table (Table 12) summarizes the results of the calculations.

Table 12: Energy from biogas utilisation and energy demand for aeration depending on the WWTP configuration and SDE pre-treatment concept

WWTP configuration	Energy from biogas	Energy for aeration	Δ
1-stage WWTP without SDE pre-treatment	13.8 kWh _{el} /PE/a	11.6 kWh/PE/a	+ 24.7 %
2-stage WWTP without SDE pre-treatment	18.4 kWh _{el} /PE/a	10.6 kWh/PE/a	+ 14.0 %
2-stage WWTP with SDE pre-treatment via nitrification	19.3 kWh _{el} /PE/a	9.3 kWh/PE/a	0.0 %
2-stage WWTP with SDE pre-treatment via nitritation	19.3 kWh _{el} /PE/a	9.6 kWh/PE/a	+ 3.2 %
2-stage WWTP with SDE pre-treatment via anammox	19.3 kWh _{el} /PE/a	9.7 kWh/PE/a	+ 4.3 %

The first stage of a 2-stage WWTP is characterized by high TSS, low sludge retention time and high sludge loading rate, which leads to a very low oxygen utilization (poor α -factor). Thus, the energy demand for aeration is correspondingly higher. By recycling the pretreated sludge dewatering effluent (SDE) to the 1st stage of a 2-stage WWTP, nitrite/nitrate contains chemically bound oxygen which allows substitution or compensation of the oxygen demand for carbon removal, reducing the energy demand for aeration. Nitrate (NO₃-N) contains the highest amount of chemically bound oxygen, which makes the SDE pre-treatment via nitrification to the most efficient treatment option. This is closely followed by the SDE pre-treatment via nitritation (+ 3.2 % energy demand for aeration) and by SDE pre-treatment via anammox (+ 4.3 % energy demand for aeration).

The SDE is characterized by high temperature (about 30 °C), high ammonium concentration (NH₄-N ca. 1 600 mg/L) and high pH value with corresponding high ammonia concentration. These factors cause inhibition of the nitrite oxidizing bacteria leading to an enrichment of nitrite and loss of nitrite oxidizing. To ensure the nitrification process in the SDE pre-treatment tank, return sludge has to be dosed to the treatment tank in order to dilute the SDE and to prevent the inhibition of NOB. This measure consequently leads to a deterioration of the oxygen transfer rate with increasing energy demand for aeration. Therefore, a stable process of nitrification in the SDE pre-treatment tank under preexisting parameters (without dilution) cannot be operated.

The nitritation-anammox-treatment shows a slightly higher energy demand for aeration compared to SDE pre-treatment via nitrification and nitritation in sidestream taking into account the entire wastewater treatment process. Full-scale applications show, that established processes like nitrification/denitrification and nitritation/denitritation are characterized by a stable process control and at the same time low maintenance compared to deammonification (Lackner et al., 2015; Jardin and Hennerkes, 2012).

Therefore, nitritation in SDE pre-treatment tank and the usage of chemically bound oxygen as an oxygen source in the 1st stage of the 2-stage WWTP appears to be the most advantageous and practicable option to save in energy demand for aeration.



3.4. N₂O emissions from biological nitrogen removal and influencing factors

For WWTPs a wide range of N₂O emissions are reported: from max. 1.1 % of the nitrogen load (Ahn et al. 2010) up to 14.6 % of the nitrogen load (Wicht and Baier, 1995) and up to 25 % of the denitrified nitrogen by Foley et al. 2010. Even if the estimated values from grab samples are in general higher as the results of online monitoring, its average is comparable with the determined emission factors of long term measurements on 8 Austrian WWTPs (Parravicini et al. 2015). The differences are probably caused not only by the chosen sampling strategy, as exemplified by Daelman et al. (2013), but also influenced by specific factors (influent characteristics, operational conditions or WWTP's configuration).

On conventional WWTPs mainly the aerated zones of the biological tanks contribute to the N₂O emissions (90% by Ye et al., 2014; Ahn et al., 2010; Lim and Kim, 2014). Since N₂O can also be produced in sewer systems (Short et al. 2014), emissions of N₂O, can be expected from clarifiers or grid tanks as well as from aerated sand traps.

With respect to the energy and cost efficiency of WWTPs, processes with low aeration energy and low COD consumption as nitrification/denitrification or partial nitrification/anammox are implemented for treatment of ammonia high concentrated sludge liquor in sidestream. Ali et al. (2016) summarised results for the N₂O emission factors of such treatment processes, with reported values for partial nitrification systems up to 6.1 % N₂O of the nitrogen load.

The biological nitrogen removal is based on oxidation of ammonia to nitrogen oxides and its further denitrification. Since the N₂O is a by-product of the nitrification and an intermediate of the denitrification, both processes can contribute to the N₂O production (and subsequently to the emission). Three important biological pathways for N₂O formation during the biological nitrogen conversion on WWTPs are postulated (Wunderlin et al., 2012):

- Hydroxylamine oxidation due to the activity of ammonia oxidising bacteria (AOB), favoured by high ammonium and low nitrite concentration in combination with high nitrogen oxidation rates;
- Nitrifier denitrification carried out by AOB under aerobic conditions and low COD loads, intensified due to lower O₂ concentrations and high nitrite concentrations;
- Incomplete heterotrophic denitrification (e.g. due to dissolved oxygen or limited organic carbon supply).

During the nitrification N₂O in wastewater treatment systems is reported to originate mainly from the nitrifier denitrification activity of AOB (Wunderlin et al., 2013). Especially at concentrations lower than 1.5 mg O₂/l, over 80 % of the N₂O production was found to originate from nitrifier denitrification (Peng et al., 2014). An increase of the concentrations to 3.5 mg/l shifts the N₂O production to the hydroxylamine oxidation pathway, however the influence of the latter on the entire N₂O production did not exceed those of the nitrifier denitrification (Peng et al., 2015). The comparable impact of the both mechanisms was confirmed by Ali et al. (2016) for granular sludge from partial nitrification/anammox reactor, furthermore the both pathways are responsible for about 70 % of the N₂O production.



In addition to the biological production, N_2O can be a product of chemical reactions. The experimental results of Law et al. (2012) could only be verified by the used mathematical model if a chemical decomposition of nitrosyl radical (NOH) was considered additionally to the nitrifier denitrification and the hydroxylamine pathway. Further, in partial nitrification systems for treatment of sludge liquor the biological production of hydroxylamine (NH_2OH) is close to the maximum due to the high ammonium concentrations and microbial activity of the AOB. This, combined with the high nitrite concentrations increases the potential of chemical N_2O formation, whereby the reaction was found to be limited by the HNO_2 concentration (Soler-Jofra et al., 2016).

For chemical phosphorus removal on WWTPs mostly iron ions (Fe^{2+}) are added in the aeration tanks and oxidised further to Fe^{3+} during the aerobic phases. Therefore, and due to the very low nitrite concentrations, the chemical production of N_2O is not expected to be relevant in aeration tanks on conventional WWTPs (Kampschreur et al., 2011). On the contrary, in sludge liquor treatment tanks employing a nitrification process, the high NO_2-N concentrations and the reduced iron in the effluent of the anaerobic sludge digester can trigger the chemical production of N_2O to a higher amount of the total N_2O emission.

Several factors impacting the N_2O production have been investigated and reported in the literature. The N_2O , as a by-product of the nitrification, is unavoidable stripped immediately after it is formed during the aeration. During the denitrification N_2O is only accumulated, when the process is limited (by dissolved oxygen or by low COD availability) and the last step of the denitrification cannot be carried out. In addition, the temperature, pH, presence of certain chemical substances can also affect the biological activity of the microorganisms during the nitrogen conversion and indirectly influence the N_2O production/emission. The dynamic of the N_2O emissions from aerated tanks in the mainstream (Parravicini et al., 2015) as well as for nitrification tanks (Ali et al., 2016; Pijuan et al., 2014; Mampaey et al., 2016) follows primarily the diurnal variation of the inflow nitrogen load, as the N_2O is a by-product of the nitrification. Nitrification tanks for SDE treatment with sequencing batch feeding (SBR) show cyclic variation of the N_2O off-gas concentrations with higher N_2O off-gas concentrations at the beginning of the aeration cycle and continuously decrease till the end of the aeration phase, irrespectively of the nitrogen load (Gustavsson and Jansen, 2011; Kampschreur et al, 2008). Furthermore, Kampschreur et al (2008) reported a similar pattern for the dissolved N_2O concentrations during the aeration phases in such tanks, but an increase during the anoxic phases, which is contrary to the observations in mainstream of WWTPs reported by Parravicini et al. (2015).

3.4.1. N-load

Since the oxidation rates during nitrification mainly depend on the ammonia and oxygen concentrations, described by the Monod kinetic, these factors seem to be the most important for a stable and complete nitrification process. The effect of NH_4-N during the nitrification was observed under laboratory conditions at low as well as at high NH_4-N concentrations (Gejlsbjerg et al., 1998; Schneider et al., 2014). During the aeration N_2O is emitted only when NH_4-N is present (Castro-Borros et al., 2016). Increased nitrogen loads lead to increased N_2O production (Lotito et al., 2012) and ammonia influ-



enced N_2O off-gas peaks (Gustavsson and Jansen, 2011). Since in conventional aeration tanks nitrate is the end product of the ammonium oxidation, the expected correlation between the nitrate concentrations and the N_2O emissions was also observed (Park et al., 2000). A linear correlation between the N_2O production rate and the $\text{NH}_4\text{-N}$ oxidation rates up to 80 mg $\text{NH}_4\text{-N}/(\text{h}\cdot\text{gVSS})$ was found (Law et al., 2011). The further exponential increase of the oxidation rates up to 300 mg $\text{NH}_4\text{-N}/(\text{h}\cdot\text{gVSS})$ was lead back on chemical breakdown of NOH (Law et al., 2012).

3.4.2. Dissolved oxygen (O_2)

The dissolved oxygen influences both, nitrification and denitrification. The emission factor for not oxygen limited nitrification remains constant due to the unavoidable stripping of the produced N_2O . Lower oxygen concentrations may cause an additional N_2O production and higher N_2O emission factors. Several authors investigated this effect, nevertheless the transition from stable to increasing N_2O factor was reported at different O_2 concentrations. In experiments with activated sludge Zheng et al. (1994) observed this effect at 1.7 mg O_2/l , Tallec et al. (2006) at 2 mg O_2/l and Peng et al. (2014) at 1 mg O_2/l . Depending on the floc size, the oxygen profile obviously plays a role as well, since for granular sludge higher transient concentration are reported (4.5 mg O_2/l by Pijuan et al., 2014).

Wang et al. (2014) reported an aerobic N_2O production in the same order of magnitude despite different O_2 concentrations (0.5-3 mg O_2/l). But concentrations below 1 mg O_2/l were found to favour the heterotrophic denitrification, so that the nitrite reduction became the main contributor to the higher N_2O emissions in their experiments. The N_2O reductase is more sensitive to oxygen than other denitrification enzymes (Otte et al., 1996), which leads to N_2O accumulation in the liquid in presence of dissolved oxygen during the anoxic phase with the consequence of higher N_2O stripping during the subsequent aeration phase. Directly after COD loading during the anoxic phase Gabarró et al. (2014) observed N_2O accumulation in partial nitrification tank due to N_2O production and lower mass transfer k_La compared to the aerated phases, correlated to insufficient denitrification process. Gabarró et al. (2014) and Mampae et al. (2016) quantifies the N_2O production of the nitritation process during the anoxic phase at 60 %-70 % of the total produced N_2O , which is emitted immediately after turning the aeration on.

3.4.3. Nitrite concentrations ($\text{NO}_2\text{-N}$)

On WWTPs, under adequate operational conditions and advantageous COD/N influent ratio for denitrification, usually no $\text{NO}_2\text{-N}$ accumulation is observed. When high ammonium concentrated SDE is treated in sidestream via nitritation and returned back into the aeration tanks of the mainstream, an increase of the $\text{NO}_2\text{-N}$ concentration in the latter cannot be excluded. Castro-Borros et al. (2016) observed approx. 5-fold higher N_2O emission during ammonium oxidation in the presence of $\text{NO}_2\text{-N}$ from SDE (up to 50 mg/l). Once the $\text{NO}_2\text{-N}$ was oxidised, the N_2O emission turned back to the level prior to SDE dosage. Step-wise increasing the concentrations up to 150 mg $\text{NO}_2\text{-N}/\text{l}$ was accompanied by further increase of the N_2O emissions as well as by inhibition of the ammonium oxidation. Higher N_2O production rate in the presence of $\text{NO}_2\text{-N}$ (up to 50 mg/l)



was observed also by Peng et al. (2015). The main pathway for N₂O production in the presence of high NO₂-N concentrations is the NH₂OH oxidation. Complete suppression of the nitrifier denitrification is reported by Law et al. (2013) in nitrification tank for sludge dewatering effluent with NO₂-N concentrations of 500 mg/l. This was later confirmed by the mathematical model of Ni et al. (2014), where the NH₂OH oxidation is the main contributor to N₂O production at very low NO₂-N concentrations (below 0.25 mmol N/l or 3.5 mg NO₂-N/l) as well as at NO₂-N concentrations over 50 mmol N/l or 700 mgNO₂-N/l. Since the free nitrous acid (FNA) concentrations are pH dependent, not only the NO₂-N concentrations, but also that of FNA may cause an inhibition. Zhou et al. (2008) found complete inhibition of the N₂O reduction at concentrations greater than 0.004 mg HNO₂-N/l in denitrifying sludge. Contrary to this, in high concentrated streams either Pijuan et al. (2014) nor Law et al. (2011) as well as Kampschreur et al. (2008) observed any inhibition at concentrations 0.065 mg HNO₂-N/l (Pijuan et al., 2014) and up to 4.6 mg HNO₂-N/l (Law et al., 2011), which was lead back on the ability of AOB to acclimate to higher FNA concentrations.

3.4.4. Insufficient COD supply

Since N₂O is an intermediate of the NO₃-N conversion to N₂, a limited or disturbed denitrification is recognised to contribute to the N₂O emissions. In addition to present oxygen during anoxic phases, also the COD supply (availability and origin) can significantly inhibit the process. As concluded by Wang et al. (2014), the lower COD content of real sludge liquor is responsible for the higher N₂O emissions compared to those, when synthetic sludge liquor (no COD content) was used. Increased N₂O production was found by Hanaki et al. (1992) at ratio of COD/NO₃-N less than 3.5 and by Park et al. (2000) at C/N ratio less than 5. Outgoing from municipal wastewater characteristics such limitation during the conventional wastewater treatment via nitrification and denitrification is rather unexpected. In the mainstream of conventional WWTPs with sufficient COD supply and complete denitrification, no N₂O production occurs during the anoxic phases and the N₂O emission was correlated to the denitrification efficiency (Parravicini et al., 2015). Thörn and Sörensson (1996) also observed higher N₂O emissions on days with lower denitrification rates. These finding are more relevant for sidestream treatment of SDE, due to the very low biodegradable COD content of the sludge liquor. In this case the reduction of NO₃-N stops at N₂O due to a competition of the participating reductases (Pan et al., 2013). Their experiments show, that a limitation of the COD leads to a lower electron supply to the N₂O reductase which subsequently causes an accumulation of N₂O. Such competition for electrons can be also observed when readily biodegradable COD is in excess, where the intensive reduction of NO₃-N also leads to accumulation of N₂O.

3.4.5. pH-value

During the nitrification an increase of the initial pH=7 to pH=8 resulted in a 3.5-fold higher N₂O emission (Law et al., 2011). During the denitrification significantly higher N₂O production at lower pH levels (pH=6.5) compared to those at pH=7.5 was observed by Hanaki et al. (1992), remaining stable for levels over pH=7.5. The N₂O accumulation found by Thörn and Sörensson (1996) was observed as soon as the pH decreased below



6.5. Based on experiments with wide variation of the pH (6-9) Pan et al. (2012) concluded, that the N_2O reductase is more sensitive to pH fluctuations compared to the other reductases of the denitrification chain. Furthermore, the accumulation at low pH levels is affected not only due to reduced N_2O reductase activity, but also due to a stronger competition between the different reductases as a result of reduced carbon oxidation rates and reduced electron supply.

Since in the aeration tanks of conventional WWTPs the pH remains relatively stable, no effect on the N_2O emissions is expected. Nevertheless, the pH changes in batch loaded sludge liquor treatment tank shows greater variation in dependence of the charging cycle, so an effect of the pH fluctuations can be expected.

3.4.6. Temperature

The temperature influences the N_2O production indirect, influencing the growth rates and the concomitant nitrogen conversion. Therefore, an increased temperature during ammonium oxidation should be accompanied by higher N_2O emissions if no other influencing factors are present. Grab samples from activated sludge tanks show no temperature influence on the N_2O emissions, neither by Thörn and Sörensson (1996) nor by Wicht and Baier (1995), despite of the wide monitored temperature range (6-25°C). Ahn et al. (2010) reported lower N_2O emissions during the winter measurements and lead these back to the lower microbial activity, but it should be mentioned, that other factors may influence the observed correlation (e.g. the chosen sampling strategy (grab samples) or diurnal variation of the loading resp. oxidation rate). Nevertheless, this results were in agreement with the experiments of Wang, X. et al. (2014), who used temperature acclimated (6 months) denitrifying sludge and demonstrated in the range 4°C to 34°C higher potential for N_2O losses at high temperature due to lower ratio of the N_2O and NO_3 reduction rates.

3.4.7. N_2O from SDE treatment in sidestream with different biological processes

The treatment of high concentrated sludge liquor (SDE) in sidestream became recently more important due to the lower oxygen demand for ammonia oxidation and lower or no biodegradable COD (bCOD) required for the nitrogen removal, when treatment processes as nitritation and subsequently denitritation or anammox are employed. The different biological processes or process combinations (nitrification/denitrification, nitritation/denitritation or partial nitritation/anammox) lead to different operational conditions as pH, NH_4^- , NO_3^- , NO_2^- -N concentrations as well as O_2 or COD concentrations. The treatment may be carried out in reactors with continuous or batch feeding, in single or in series-connected reactors.

The combination of the influencing factors (high ammonium and nitrite concentration, batch loading, low COD, high temperature and high pH fluctuations) leads to higher N_2O production and emission from nitritation tanks compared to other processes (nitrification/denitrification, nitritation/anammox).

In full scale partial nitritation systems the N_2O emission factor reaches from 1.7 to 6.6 % of the nitrogen load, which corresponds to 3.4-11.2% of the oxidised NH_4 -N (Kampschreur et al., 2008; Desloover et al., 2011, cited in Pijuan et al., 2014). No or lower



N₂O emissions are reported from reactors, where the nitrification is combined with a further process for nitrogen removal (e.g. from anammox reactor or nitrification/anammox reactor) compared to single nitrification reactors (Desloover et al., 2011; Kampschreur et al., 2008). Mampaey et al. (2016) attributed this to the higher NO₂-N concentrations. N₂O emissions from anammox reactor of two-stage processes are probably caused by nitrifiers, replaced with the effluent of the nitrification reactor to the anammox reactor (Kampschreur et al., 2008).

Rodriguez-Caballero et al. (2013) found considerably lower N₂O emissions from nitrification processes compared to those from nitrification (0.54 % N₂O-N/N_{converted} resp. 1.22 %N₂O-N/N_{converted}).

3.4.8. Mitigation measures

During in aeration tanks of the mainstream longer denitrification phases and shorter aeration phases reduce the N₂O emission (Kimochi et al., 1998; Gejlsbjerg et al., 1998), in SDE treatment tanks longer anoxic phases increase the N₂O emission due to increased N₂O formation (Mampaey et al., 2016; Gustavsson and Jansen, 2011; Kampschreur et al., 2008). When continuous instead of intermittent aeration is employed, the emission factor decreases (Joss et al., 2009). Thereby the O₂ concentrations should exceed 1 mg/l to prevent rigorous increase of the N₂O emission factor (up to 18.5 % at 0.6 mg O₂/l).

The feeding strategy is another parameter, influencing the N₂O emissions. Pijuan et al. (2014) observed an increase of the N₂O emission factor from 2.2 % to 19.3 % when the operation was changed from continuous to SBR mode. Fux et al. (2006) as well as Law et al. (2011) also reported better results for SBR with continuous feeding or slow-feeding. Maintaining the pH level between 6 and 7 combined with slow-feeding was reported to reduce the N₂O emissions by factor 4. This low ammonia fluctuation feeding strategy is in line with the findings of Law et al. (2012), that lower ammonia oxidation rates reduce the N₂O production.



4. Materials and Methods

4.1. Description of tools

4.1.1. Collecting and visualization of data with iTUWmon

Data analysis and modelling of parameters concerning the WWTP Kirchbichl is based on automatic transfer of operational data, appended into data structures on the central data aggregation workstation. High temporal resolution raw datasets are available, offering a time grid of 3 minutes between two individual measurements. The synchronization-task is repeated every three hours for keeping the network load requirements relatively low. A total of approximately 1,100 data channels are incorporated into the central data storage, which then are available for name-based data export into a common, text based data exchange format for detailed investigation.

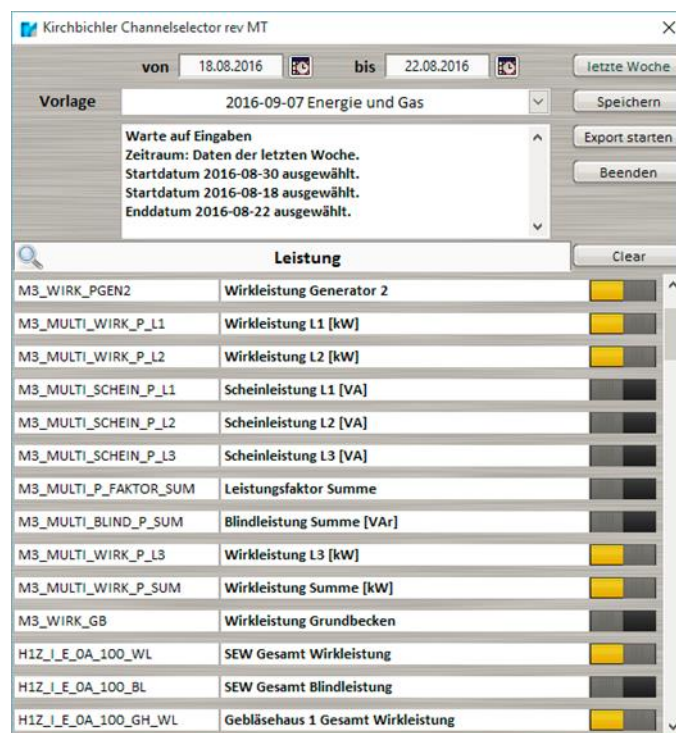


Figure 21: Software tool for data channel selection

A software tool was created, allowing for export of WWTP's data channels of interest. The user interface of the tool is depicted in Figure 21. Based on control system channel names on site, the user is able assemble several corresponding datasets of interest and save the selections for repeated, recurring analysis tasks in the future, including complete up-to-date measurement values.

4.1.2. Methods for mass balances

The comparison of the both scenarios (prior and after implementing the nitrification of SDE in sidestream) will be based on mass balances and key performance parameters, mainly derived from WWTP's data. Since the collected data could be affected by systematic or measurement errors due to sampling, chemical analysis or measurement equipment, a validation of the used datasets should be undertaken prior fitting these into the simulation models. A reliable method for verification of WWTP's data is mass balancing for selected parameters as COD, TN and TP (Nowak *et al.*, 1999). The balancing tool eDAB (www.edab.at) has been applied for the calculations and visualisation of the results.

An essential step of each balancing process is the determination of the system borders. For total balances the borders of the WWTP were chosen to be the cadastral borders, so that all sub-processes of the wastewater treatment (primary clarification, biological treatment, sludge storage and dewatering, sludge treatment, secondary clarifying) are considered and the WWTP is assumed to be a "black box". In addition to the total balances, detailed balances for single WWTP's modules are possible if the corresponding in- and output streams are determined. A "module" is defined as a single tank (e.g. sand trap, sludge storage tanks, digester) or group of tanks where the streams are stored or treated (e.g. sludge dewatering station, combination of aeration tanks and clarifiers). A description of the balancing method and the used equations is following given. To simplify the demonstration of the balancing method for a conventional WWTP, the mechanical treatment and the biological treatment are combined in one module, "biological step", as well as the sludge thickeners, digester, sludge dewatering and SDE treatment tank in the module "sludge treatment".

Assuming that for long monitoring period (i.e. 2-3 times the sludge age) the storage processes can be neglected due to their lower relevance compared to the total loads turnover on the WWTP, under steady state conditions the difference of all in- and output loads should amount to zero (Svardal *et al.*, 1998). A load is determined by its flow and concentration for a defined time unit (kg/d or m³/d). In case of closed mass balance all loads can be calculated by mean of measured values, which allows a data plausibility check, otherwise the balance is open. In general, more sampling points, higher intensity of the sampling and quality of the analyses increase the quality of the mass balance. A negative balancing gap shows a higher output compared to the input loads, a positive one corresponds to higher input as output loads. Gaps of $\pm 10\%$ were accepted, otherwise additional measurements and calculations were undertaken to clarify the reason and to correct the deviation.

Since the water flow balance is based primarily on measurements, no further estimations are usually required. Much elaborated are mass balances for chemical compounds. The incoming substances remain stable or are transformed due to the treatment process and discharged to the environment via the effluent, the dewatered sludge and the off-gas. Due to the limited data for chemical analyses on conventional WWTPs, especially for internal streams, it is possible to balance the WWTP with lower effort, provided that analyses for the total parameters such as total Phosphorus (TP), total Nitrogen (TN), chemical oxygen demand (COD) are available.



Mass balance for MLSS

The calculation of the MLSS-loads (mixed liquor suspended solids) is based on the water flow (m^3/d) and measurements of its MLSS concentrations (kg/m^3). Since on WWTPs the MLSS concentrations of the influent and effluent are usually not measured, for the total balance only the load of the dewatered sludge can be calculated. For detailed balances for modules with biological conversion, the output MLSS loads cannot be determined easily due to the not quantifiable biomass growth and decay. An easy mass balance of biological process is possible only for the anaerobic digesters. MLSS mass balances were used mainly to check data availability of mechanical sludge treatment steps. The mass balances for all thickeners and sludge dewatering stations are based on the assumption, that biological activity can be neglected. The only mechanical process is the increasing of the MLSS-concentration in outgoing flows, so the MLSS load of the sludge stream (in- and output load) is equal. The quantified and verified MLSS-loads are particularly used for calculation of the total nitrogen, total phosphorus and COD balances.

Total phosphorus mass balance

In the influent of WWTPs phosphorus is present in organic bound and dissolved form, which indeed underlie transformation processes during the wastewater treatment but remain, as shown in Figure 22, in liquid or solid streams and can be determined using the analytical parameter total phosphorus (TP).

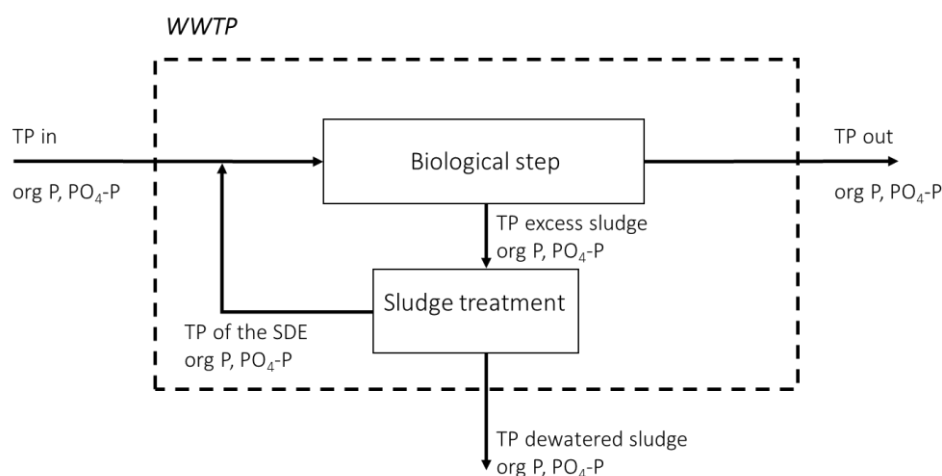


Figure 22: Flow scheme for total phosphorus balance on WWTPs

Derived from the balance definition, the following mathematical equation was used for the total phosphorus balance:

$$Q_{in} * TP_{in} = Q_{out} * TP_{out} + Q_{DS} * TP_{DS}$$

Since on the WWTP Kirchbichl the concentration of TP in influent and effluent was regularly measured, but not in the dewatered sludge, further measurements were necessary to estimate these TP-load. Advantageous for such estimations are the ratios of the particulate compounds (Pp, Np, CODp) and the content of volatile suspended solids (VSS) due to their lower variation compared to the ratios (TP, TN, COD)/MLSS, especially when inorganic additives are used for dewatering (Svardal et al., 1998). Therefore, the ratio



$\frac{P_{pDS}}{VSS_{DS}} = \frac{TP_{DS} - PO_4 - P_{SDE}}{VSS_{DS}}$ was evaluated. Once the TP-load of the dewatered sludge was determined, the WWTP was balanced according to the transformed equation:

$$Q_{in} * TP_{in} = Q_{out} * TP_{out} + MLSS_{DS,load} * \frac{TP_{DS} - PO_4 - P_{SDE}}{VSS_{DS}} * \frac{VSS_{DS}}{MLSS_{DS}} + Q_{DS} * PO_4 - P_{SDE}$$

The single modules of the WWTP are balanced by the same procedure. The mass balance of the anaerobic digester is used for verification of the measurements, since the phosphorus remains in the sludge despite of the biological conversion. The TP balance is a redundant and reliable instrument for verification of total solid loads when the ratios Pp/VSS of the internal loads are known.

Total nitrogen mass balance

The legally required nitrogen elimination for WWTPs with a capacity of more than 5.000 PE in Austria (at least 70 %, 1.AEV,1996) can only be reached, when the inflow nitrogen is converted to nitrogen gas by denitrification. Since N_2 is released to the atmosphere mainly from the biological treatment tanks and is not usually measured, the total nitrogen mass balance of a WWTP is open (Figure 23). If the nitrogen in the dewatered sludge is estimated based on the Np/VSS ratio and the MLSS-load, the load of denitrified nitrogen can be derived from the transformed balance equation:

$$N_2 = Q_{in} * TN_{in} - Q_{out} * TN_{out} - MLSS_{DS,load} * \frac{(TN_{DS} - NH_4 - N_{SDE})}{VSS_{DS}} * \frac{VSS_{DS}}{MLSS_{DS}} - Q_{DS} * NH_4 - N_{SDE}$$

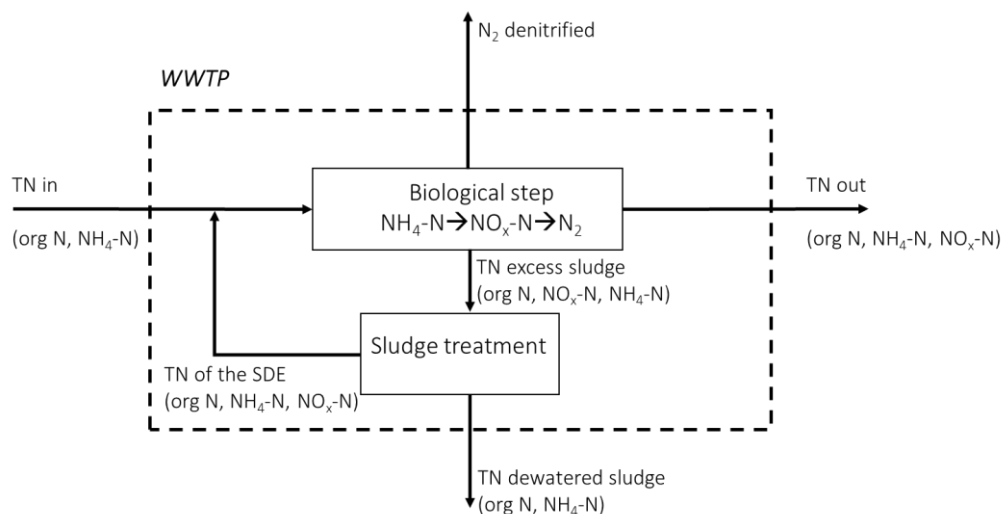


Figure 23: Flow scheme for total nitrogen balance on WWTPs

Similar as the TP sludge loads, the TN sludge loads are estimated based on the Np/VSS ratio and the MLSS-loads of the internal sludge streams for the detailed mass balance. The mass balance of the digester is used for verification of the measurements, since the nitrogen remains in the sludge despite of the biological conversion.

COD mass balance

On WWTPs the inflow COD passes through the biological step and is emitted with the effluent, converted to biomass (excess sludge) or oxidised to CO_2 and emitted to the atmosphere in the biological step or to CO_2 and CH_4 in case of sludge digestion, re-



spectively (Figure 24). The COD mass balance is the most elaborate since additional calculations and much more data of different origin are required. Nevertheless, it can be used successfully to prove the quality of measured data. For the total COD balance, a preliminary estimation of partial loads is necessary (COD_C , COD_{CH_4}).

The balance of the biological step is calculated by means of following equation:

$$Q_{in} * COD_{in} - Q_{out} * COD_{out} = MLSS_{ES,load} * \frac{COD_{ES} - COD_{out}}{VSS_{ES}} * \frac{VSS_{ES}}{MLSS_{ES}} + COD_C$$

As for the TP-loads, also the COD-loads in sludge streams are estimated based on the COD_p/VSS ratio and the MLSS-loads.

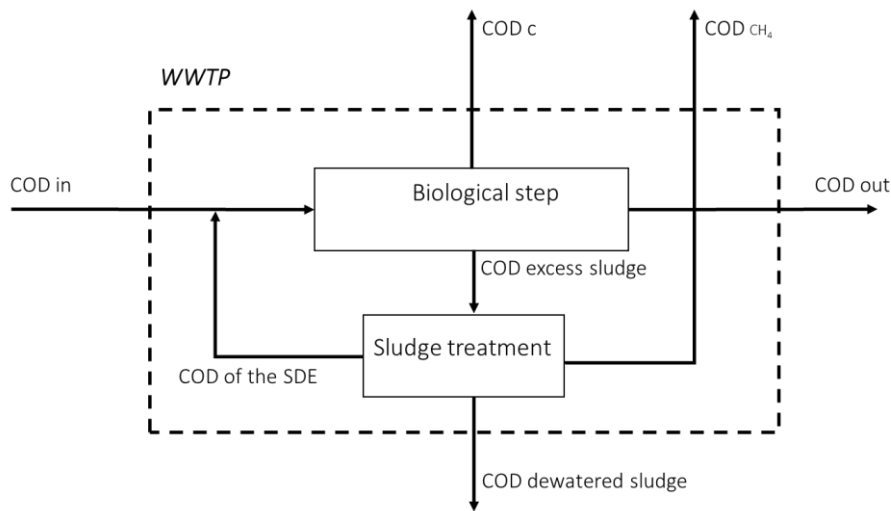


Figure 24: Flow scheme for COD balance on WWTPs

The degradation of carbonaceous organic matter (COD_C) occurs under aerobic conditions during its oxygenation, as well as anoxic while nitrogen oxides play the role of electron acceptors. Since the bacterial respiration is commonly not measured on WWTPs, the aerobic degradation of COD can be calculated based on the supplied oxygen to the biological step as follows:

$$COD_C = OU - OU_{nitri} + OU_{deni}$$

The oxygen consumption of each aerated tank is calculated by means of:

$$OU = E_{aeration} * OTR * \frac{C_s - C_x}{C_s}$$

OU oxygen consumption, kg O_2 /d

$E_{aeration}$ energy for aeration, kWh/d

OTR oxygen transfer rate, kg O_2 /kWh

c_s saturation concentration for O_2 , mg/l

c_x measured O_2 concentration, mg/l

Data for energy consumption as well as for the oxygen concentration in all aerated tanks on the investigated WWTPs are usually available.

The OTR value was iterated for the interval 1.3 kg O_2 /kWh to 2.8 kg O_2 /kWh automatically by the tool eDAB until the gap of the COD balance for the investigated aeration tank reached its minimum. Since the calculated oxygen consumption includes also the oxy-



gen consumed for nitritation or nitrification, the latter one has to be subtracted. The load of oxidised nitrogen was calculated by means of:

$$N_{\text{oxidised}} = Q_{\text{in}} * TKN_{\text{in}} - Q_{\text{out}} * TKN_{\text{out}} - \left(MLSS_{\text{DS,load}} * \frac{TKN_{\text{DS}} - NH_4\text{-}N_{\text{SDE}}}{VSS_{\text{DS}}} * \frac{VSS_{\text{DS}}}{MLSS_{\text{DS}}} + Q_{\text{DS}} * NH_4\text{-}N_{\text{SDE}} \right)$$

The nitrification in all tanks on the investigated WWTP Kirchbichl was assumed to be completely, excepting the oxidation process in the SDE treatment tank, operated via nitritation after the process modification. The load of oxidised nitrogen was calculated by means of:

$$OU_{\text{nitri}} = f_{\text{oxygen}} * N_{\text{oxidised}}$$

f_{oxygen} factor for O₂ consumption (4.3 g O₂/g NH₄-N and 3.2 g O₂/g NH₄-N for nitrification and nitritation, respectively)

Since heterotrophic microorganisms participate also to the anoxic COD degradation, the COD respiration is calculated by multiplying the reduced nitrogen load (denitrified as well as denitrified) by a factor for specific COD consumption (f):

$$OU_{\text{deni}} = f * N_{\text{reduced}}$$

f factor for COD consumption (2.86 g COD/g NO₃-N resp. 1.72 g COD/g NO₂-N)

In case the excess sludge is digested, the COD of the produced methane must also be considered as output pathway. The in- and output streams for the module sludge treatment are usually measured or can be calculated, so that a closed COD balance for the digesters is possible. Since the volume of the produced biogas and the content of CO₂ are also measured, the CH₄ load over the produced biogas can be determined. An additional method to calculate or prove the CH₄ production is based on the produced electricity, measured or derived from the working hours of the implemented combined heat and power units (CHP), empirical values regarding the heat value of biogas (1 m³_{i,N} equals 10 kWh) and assuming CHP-efficiency of 25-30 %. In both cases the CH₄-load is multiplied by its COD equivalent (4 kg COD/kg CH₄ or 2.86 kg O₂/m³_{i,N} CH₄).

$$COD_{\text{CH}_4} = 2.86 * CH_4 \text{ load}$$

Once the additional calculations are finalised, the total balance of the WWTP can be closed.



4.2. Analytical methods

WWTP's data regarding wastewater, sludge and effluent characteristics were provided by the plant operator and used for the mass balances. Since measurements of the internal streams are not required on WWTPs, additional measurements were undertaken to compensate the lack of data for important internal water flows and sludge streams. The samplings points were chosen in conformity with the WWTP's design in order to obtain the required information for balancing of the single modules by keeping a low additional analytical effort. The analyses of the returned sludge and of the effluent of the 1st stage for example provide information on sludge loads up to the thickener effluent including the returned streams to the 1st stage. Analyses of the digester effluent were informative for the streams dewatered sludge and sludge dewatering effluent.

Once weekly a set of samples were delivered to the laboratory of the Research Centre for Water Quality (TU Wien) and analysed for the selected parameters using the methods summarised in Figure 25.

Parameter	Analytical method	Sampling point								
		Effluent 1st stage	RAS 1 st stage	RAS 2 nd stage	Recycle line digester 1	Recycle line digester 2	mechanical thickened sludge	dewatered sludge	sludge dewatering effluent (SDE)	Effluent SDE treatment tank
COD	DIN 38409-43	+	+	+	+	+	+			
TKN	Standard Methods 4500-N org.		+	+	+	+	+			
TP	DIN EN ISO 6878 adopted Methode (microw ave digestion)		+	+	+	+	+			
NO ₂ -N, NO ₃ -N	DIN EN ISO 13395	+							+	+
NH ₄ -N	DIN EN ISO 11732	+			+	+			+	+
PO ₄ -P	DIN EN ISO 6878	+			+	+			+	+
MLSS	DIN 38409-1		+	+	+	+	+	+		
VSS	DIN 38409-2		+	+	+	+	+	+		

Figure 25: List of the sampling points for additional measurements as well as of the applied analytical methods

4.3. Equipment for N₂O measurement

The off-gas from the aerated tanks on the WWTP was collected using a gastight, wooden floating hood with dimensions 1 000 x 1 000 x 500 mm and thickness of 9 mm. The inner sides were equipped with 5 cm tick Styrodur plates and the volume of the headspace varied between 0.28 m³ when the aeration was turned off and 0.31 m³ when aerated. The floating hood was positioned in the middle of the aerated surface in the monitored aerated tank and fixed using four ropes (as shown on Figure 26). The off-gas was transferred from the floating hood headspace to the measurement units via transparent polyvinyl chloride hoses with diameter of 8 mm, thickness of 1.5 mm and length of 25 m. Prior to the measurement units, the off-gas was passed through a unit for humidity and dust removal (PSS-5, M&C Tech group, Germany) with integrated pump (capacity of 120 l/h). After the off-gas treatment, the stream was divided in two equal sub-streams and passed through parallel measurement units for N₂O and CO₂. Both, N₂O and CO₂ were measured continuously and the data stored in a minute cycle. An off-gas stream of 60 l/h was pumped through an N₂O infrared analyser (Thermo Scientific™ Model 46i Nitrous Oxide Analyser with range 0.02-50 ppmv). The remaining stream of 60 l/h passed the CO₂ infrared measurement unit (NDIR 7000, Fa. SAXON with a range of 0-10 vol.-%). The on-line measurement of CO₂ allows a continuous check of the airtightness of the measurement set-up. Decreasing CO₂ concentrations during the anoxic phases indicate that ambient air penetrates in the system and dilutes the N₂O off-gas concentration. The exhaust air of both measuring units is merged and returned back to the floating hood through a second hose, situated at the opposite corner of the take-off hose as shown in Figure 26. The recirculation is necessary due to the continuous uptake of off-gas from the headspace of the floating hood. At intermittent aerated fields, when the aeration is turned off for longer period (>2 h) for denitrification, this in circle driven off-gas stream stabilises the pressure in the headspace preventing damages of the equipment from withdrawn water. All connections to the floating hood are performed with self-sealing valves to avoid inlet of ambient air.

Due to higher N₂O off-gas concentrations, exceeding the upper measurement limit of 50 ppmv, the off-gas was diluted in a defined ratio with ambient air prior passing the off-gas treatment unit.



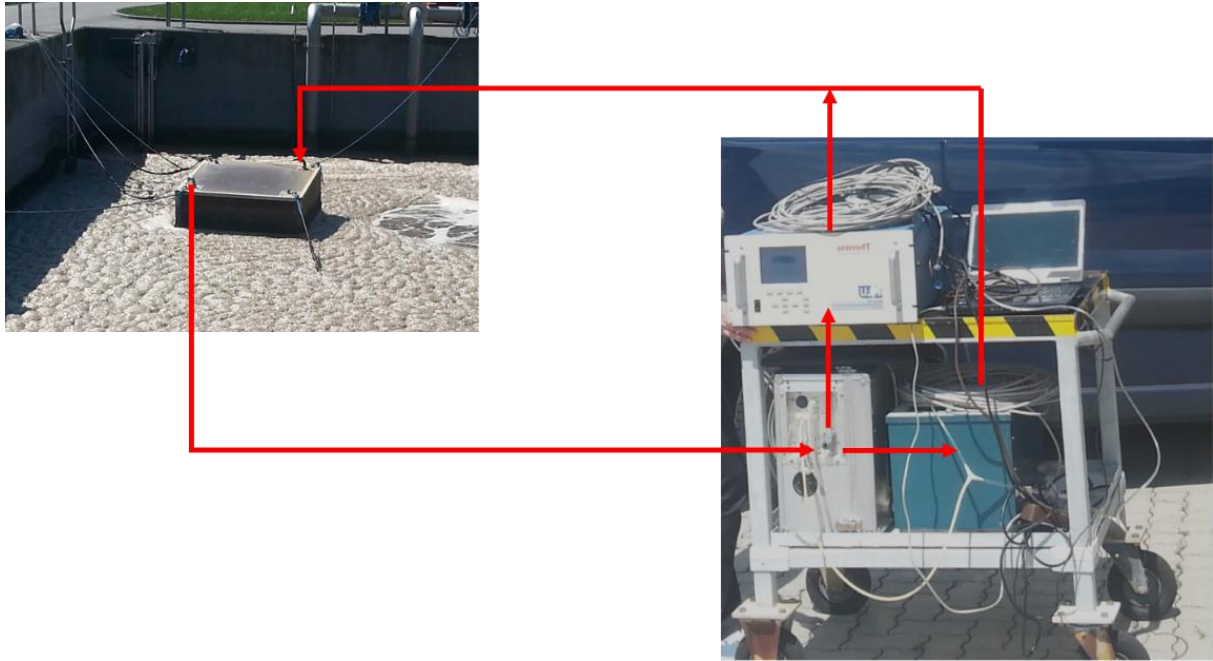


Figure 26: Set up for continuous N₂O off-gas measurement

The CO₂ analyser was checked under laboratory conditions prior starting the measurement campaign and once weekly on-site during the measurement campaigns using a GFM 410 unit for O₂ and CO₂ control (Gas Data Ltd). Tests with gas mixture (N₂ and CO₂) under laboratory conditions and varying CO₂ concentration (0-10 vol.-%) were carried out to determine the cross-sensitivity of the N₂O measurement for CO₂. The N₂O analyser was calibrated prior to the measurement campaign and the base line controlled once weekly with N₂.

A pH sensor was positioned in the activated sludge tank near the floating hood and the values were transferred to iUmon in minute intervals. Data for O₂ concentrations in the aerated tanks were obtained from the WWTP in 3-minute intervals.

4.4. Data management

4.4.1. Calculation of N₂O-emissions and emission factors

Based on results of previous N₂O measurement campaigns (Parravicini et al., 2015) carried out by the team of the Research Centre for water quality (TU Wien) it can be postulated, that the N₂O concentration of the off-gas collected from the middle of a monitored aerated field is representative for the off-gas from the entire carousel aerated tank. Moreover, the N₂O emissions of parallel, not monitored aeration tanks (as the second stage on WWTP Kirchbichl) can be estimated based on the supplied airflow in each tank and the measurement results from one of them, so far loading and operational conditions are comparable.

For not aerated phases and not aerated zones N₂O can be emitted to the ambient through the water surface following the gas equilibrium. Results from previous measurements show, that the N₂O emission in case the aeration system is turned off can be neglected (<0,1 % of those when aerated), so these were not considered.

Since on the monitored WWTP no data for the airflow are available, but the air supplying system is equipped with air diffusers and controlled by compressors frequency, the supplied air volume can be estimated based on manufacturers data and available data for the compressors frequency. An additional correction by means of meteorological data (pressure, temperature and atmospheric humidity) is necessary to convert the estimated airflow to standard conditions (T=273.15°K and p=1013 mbar).

The volume of the off-gas will be derived from the airflow balance of each investigated tank. Ambient air is compressed and supplied by air diffusor system on the bottom of the tank. Concerning the different solubility of the gases as well as the biological utilization of O₂ and stripping of CO₂ from the treated wastewater, the volume of the off-gas differs slightly from the volume of the supplied air. The flow of inert gases (mainly N₂ and Ar) is assumed to remains constant, so the exhaust air flow can be estimated by the following equation:

$$Q_{\text{air,in}} * (p_{\text{N}_2,\text{in}} + p_{\text{Ar},\text{in}}) = Q_{\text{air,out}} * (p_{\text{N}_2,\text{out}} + p_{\text{Ar},\text{out}}), \text{ (Frey, 1989)}$$

The partial pressure of the inert gases was not measured, but the CO₂ and O₂ concentration in the off-gas, so the off-gas airflow can be calculated by means of the transformed equation:

$$Q_{\text{air,out}} = Q_{\text{air,in}} * \frac{(p_{\text{N}_2,\text{in}} + p_{\text{Ar},\text{in}})}{(100 - p_{\text{O}_2,\text{out}} - p_{\text{CO}_2,\text{out}})}$$

$Q_{\text{air,out}}$ off-gas airflow, m³/min

$Q_{\text{air,in}}$ aeration airflow, m³/min

$p_{\text{N}_2,\text{in}}$ partial pressure of nitrogen gas in ambient air, %

$p_{\text{Ar},\text{in}}$ partial pressure of argon in ambient air, %

$p_{\text{O}_2,\text{out}}$ partial pressure of oxygen in the off-gas, %

$p_{\text{CO}_2,\text{out}}$ partial pressure of carbon dioxide in the off-gas, %

Taking this into account, the N₂O emission during the aerated phases can be calculated as the product of the N₂O off-gas concentration and the off-gas volume from the



erated tanks. Due to the utilization of ambient air for aeration, the measured N₂O concentration in the off-gas should be corrected by the ambient N₂O concentration.

$$N_2O_e = \sum_{i=1}^j (N_2O_i - N_2O_{i,aa}) * Q_{i,air,out}$$

N₂O_e N₂O emission from activated sludge tank, mg/min

N₂O_i measured N₂O off-gas concentration, mg/m³

N₂O_{i,aa} N₂O concentration of ambient air, mg/m³

Q_{i,air,out} off-gas flowrate (for standard conditions), m³/min

The daily N₂O emission for each activated sludge tank is defined as the amount of the calculated N₂O emissions on minute basis for a period of 24 h. For each treatment stage the daily N₂O emissions of all tanks will be averaged and normalised by the nitrogen load of the WWTP to calculate the corresponding N₂O emission factors. The average N₂O emission factor for the whole campaign results from the sum of the emission factors of all treatment stages.



5. Case study 5 – Results and discussion

5.1. 2-stage WWTP plant Kirchbichl

The WWTP Kirchbichl was built in the time period of 1985 – 1987 with a capacity of 83 300 PE. The plant was designed as a two-stage bio-aeration process on the basis of the adsorption-bio-aeration process technology (A-B-process, shown in Figure 28) using the aerated grit chamber as adsorption stage. The aerated grit chamber combined with the intermediate clarifier (currently used as intermediate clarifier I) was operated as a high loaded biological treatment stage. The accruing biomass was discharged as excess sludge and anaerobically digested for biogas production. The aim of the A-B process was to decrease the COD-loading rate in the B-stage due to a biological degradation in the aerated grit chamber. Therefore, a part of accruing sludge from the intermediate clarifier was returned and merged with the incoming wastewater influent. The adsorption of incoming carbon in the A-stage reduced the subsequent organic load of the B-stage. In further consequence, this led to a reduced oxygen demand associated with decreasing energy consumption for aeration. The described adsorption of carbon in the sludge of the A-stage caused a COD-enrichment. This higher amount of COD in the excess sludge from A-stage compared to those of conventional preliminary clarifiers led to a higher biogas yield in the digester. Even then, the implementation of the innovative A-B process was a significant contribution towards a more energy efficient wastewater treatment.

Between 1995 and 1996 the plant was additionally equipped with a composting plant for the accruing digested sludge.



Figure 27: State of the WWTP Kirchbichl in the year 2000



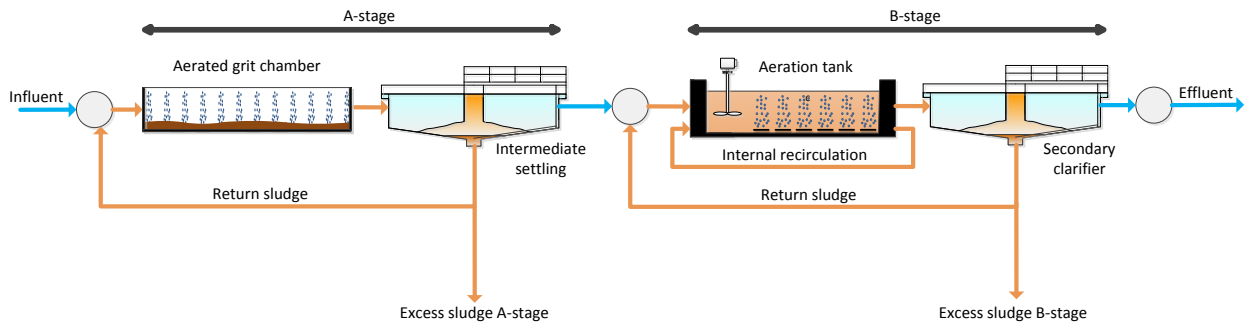


Figure 28: Flow scheme of A-B-process WWTP

As a result of increasing legal requirements concerning the removal of nitrogen and phosphorus, a further extension of the WWTP was necessary. As a part of the implementation of the new recommendations pertaining to the stricter effluent quality, the capacity of the plant was increased from 83 300 PE to 100 000 PE. The wastewater line, sludge line as well as the composting plant were affected by those measures. During the expansion two aeration tanks for a high-loaded biological stage (5, Figure 30) and a second intermediate clarifier (Intermediate clarifier II, 7, Figure 30) were constructed. Additionally, a non-aerated tank (8, Figure 30) was established and used as pre-denitrification unit with the aim to replace the denitrification volume of the intermittent aerated tanks (9, Figure 30). The thereby achieved volume in the previously intermittent aerated tank (B-stage) is now available for full aeration if necessary and increases finally the capacity of both, nitrification and COD removal.

Due to both, an increasing amount of co-substrates and a higher amount of excess sludge an upgrading of the current digester capacity was indispensable. Therefore, a second digester was implemented in the sludge line (17, Figure 31). The following aerial photo shows the WWTP Kirchbichl after the described expansion measures.



Figure 29: State of the WWTP Kirchbichl in the year 2015

5.1.1. Parameters of the WWTP

The following Table 13 shows the main parameters of the WWTP Kirchbichl.

Table 13: Plant characteristic

Plant capacity:	max. 100 000 PE	
Volume of wastewater:	Dry weather flow: max. 410 L/s,	max. 24 000 m ³ /d
	Storm water flow: max. 720 L/s	
Amount of sludge:	approx. 4 tons TS/d	
Biogas production:	approx. 2 000 m ³ /d	
Required effluent quality:	BOD ₅ < 15 mg/L	> 95 % reduction
	COD < 75 mg/L	> 85 % reduction
	TOC < 25 mg/L	> 85 % reduction
	NH ₄ -N < 5 mg/L (Temp. > 8 °C)	
	TP < 1 mg/L	
	TKN > 70 % reduction (Temp. > 12 °C)	



Table 14: Design data of the plant components

Plant component		No. of units		Total
Lifting equipment for influent	Spiral pump	3	240 L/s	720 L/s
Grid chamber	Gap width approx. 6 mm	2		720 L/s
Aerated sand trap	Sand trap with grease trap included	1	650 m ³	650 m ³
Aeration tank 1 st stage	Rectangular tank	2	750 m ³	1 500 m ³
Intermediate clarifier	Circular sedimentation tank	2	2 350 m ³	4 700 m ³
Return sludge 1 st stage	Spiral pump	2	260 L/s	520 L/s
Aeration tank 2 nd stage	Preceded denitrification tank	1	1 350 m ³	1 350 m ³
	Activated sludge tank with circulated flow	3	1 930 m ³	5 800 m ³
Secondary clarifier	Circular sedimentation tank	3	2 700 m ³	8 100 m ³
Return sludge 2 nd stage	Spiral pump	3	220 L/s	660 L/s
Excess sludge	Mechanical excess sludge dewatering	2	50 m ³ /h	100 m ³ /h
	Pre-thickener	1	700 m ³	700 m ³
Anaerobic stabilization	Digester	2	2 300 m ³	4 600 m ³
	Post-thickener	1	800 m ³	800 m ³
Sludge dewatering	Centrifuge	2	600 kg TS/h	1 200 kg TS/h
SDE treatment tank	Rectangular tank	1	580 m ³	580 m ³
Biogas	Biogas tank	1	1 000 m ³	1 000 m ³
	Block heat and power station	2	250 kW _{el}	500 kW _{el}
Composting	Composting plant			10 000 m ³



5.1.2. Flow scheme of the plant and description of the partial flows

In this chapter the wastewater and sludge treatment lines will be shortly described.

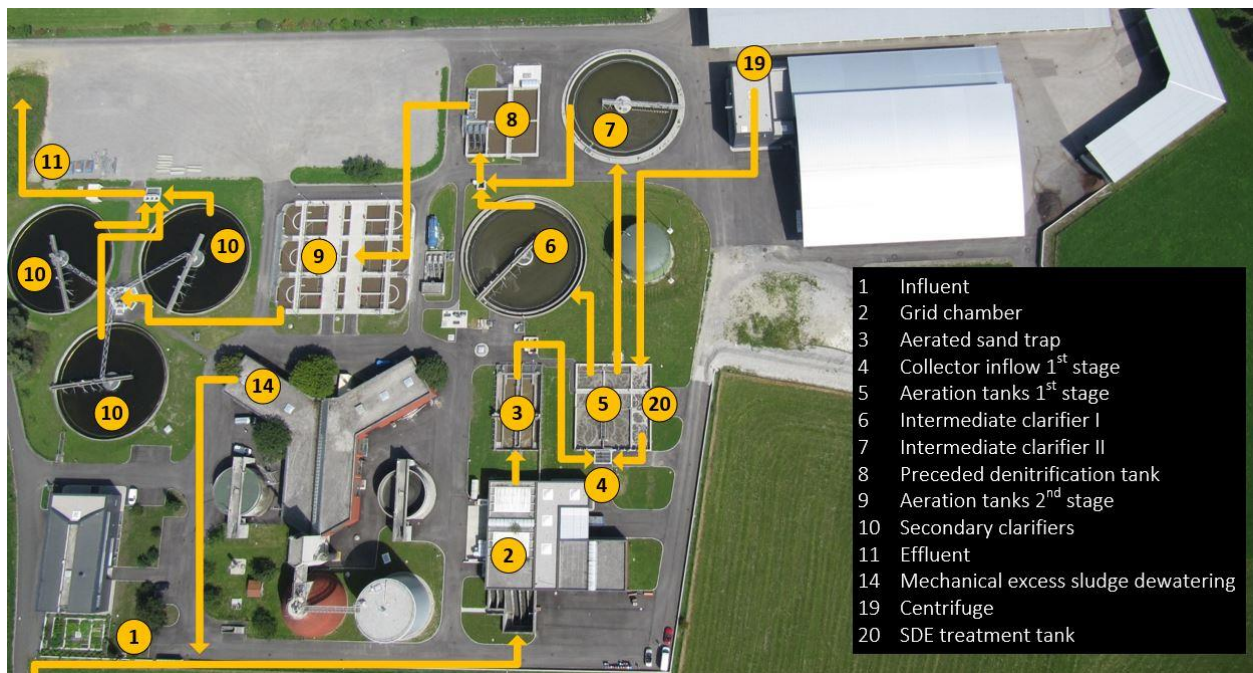


Figure 30: Water flow scheme for WWTP Kirchbichl

Wastewater treatment line

The wastewater of the associated municipalities of Kirchbichl and Wörgl (about 17 000 m³/d) is collected via mixed sewer system with partial discharge of diluted wastewater over rain overflow basins in case of storm. Daily mixed sampling from the influent (1, Figure 30) are collected prior discharging the supernatant from the mechanical excess sludge dewatering (14, Figure 30) to the influent and analysed for selected parameters. Since the influent contains pre-treated wastewater and biomass from indirect dischargers, biological processes in the sewer system are expected and confirmed by the higher NO₃-N concentration and lower COD and BOD concentration compared to the influent characteristics of other municipal WWTPs. Due to the lower altitude of the WWTP, after the sampling point for daily mixed samples the influent needs to be pumped 5 m to reach the mechanical treatment step.

The wastewater passes subsequently the 6 mm bar space screens, situated in the grid chamber (2, Figure 30). The screenings are washed and compressed by an automatic screenings washer and press unit and sent to further treatment. In the last part of the mechanical treatment, the aerated sand trap (3, Figure 30) fat and sand are removed. The COD concentration of grab samples in addition to the large volume of the sand traps (corresponding to a retention time of 55 min) let assume partial degradation of organic matter, which is considered disadvantageous for denitrification as well as biogas production.

The effluent of the sand trap is piped to the biological step of the WWTP, designed with two stages- a high- and low-load stage, each equipped with aeration tanks and their corresponding clarifiers. The influent of the high-loaded 1st stage consists of the me-



chanically treated wastewater and returned activated sludge from the intermediate clarifiers as well as the treated effluent from the sludge dewatering, mixed together in the collector (4, Figure 30). The water-sludge mixture is split to the two reaction tanks (5, Figure 30) of the 1st stage, equipped with air diffusers for oxygen supply if aeration is required. Subsequently the mixture is piped to the both intermediate clarifiers (6 and 7, Figure 30), where the biomass settles and is withdrawn. Each of the reaction tanks is connected to a corresponding clarifier, so the two lines can be processed independently from each other. The excess sludge is withdrawn from the returned activated sludge pipe.

The effluent of both intermediate clarifiers is merged and mixed with the returned activated sludge from the secondary clarifiers and lifted into the pre-denitrification tank of the 2nd stage (8, Figure 30). Subsequently the stream is split to the three parallel carousel tanks of the low-loaded 2nd stage (9, Figure 30), also equipped with air diffuser systems for intermittent nitrification. The effluent of the aerated tanks is merged again prior loading the three secondary clarifiers (10, Figure 30). The settled biomass is mainly returned prior the denitrification tank. The effluent of the WWTP (11, Figure 30) is discharged to the receiving water. If required, a part of the effluent can be returned to the 1st stage to obtain higher removal rate and better effluent's quality.

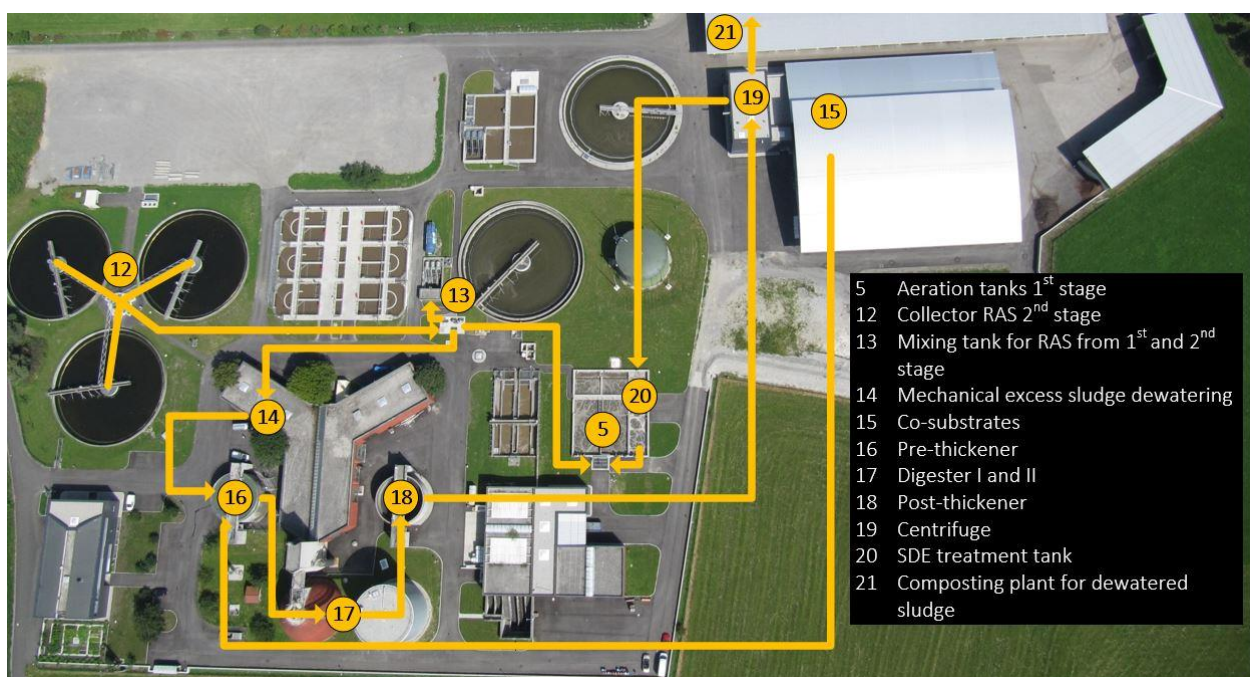


Figure 31: Sludge flow scheme for WWTP Kirchbichl

Sludge treatment line

As already mentioned, the excess sludge is withdrawn only from the returned activated sludge of the 1st stage (RAS, 1st stage). The splitting of the streams into excess sludge and RAS 1st stage occurs in a separate tank (13, Figure 31), where the settled sludge of the 2nd stage is also returned. In accordance to the HYBRID[®] concept, a partial stream of the excess sludge 1st stage (SK 1) is mixed with the RAS 2nd stage and finally mixed with the effluent of the intermediate clarifiers prior the denitrification tank. Part of RAS 2nd stage (SK 2) is streamed to the 1st stage via the SDE treatment tank (20, Figure

30) prior to the implementation of nitrification. After the operational change this stream was redirected to the 1st stage, so the influent of the SDE treatment tank consists only of the SDE.

The excess sludge is directed to the mechanical dewatering station (14, Figure 31), where a dry matter concentration of 5% is reached. It is piped subsequently to a pre-thickener (16, Figure 31), where co-substrates are dosed and mixed for the further treatment (15, Figure 31). The supernatant of the pre-thickener and the mechanical excess sludge dewatering station are returned to the influent and mixed prior the screens (14, Figure 30). The both parallel digesters (17, Figure 31) are operated with a sludge retention time of 30 d at temperature of 37°C. The influent mixture is warmed up to this process temperature prior feeding. The digested sludge has a dry matter content of 3% and is piped to the post-thickener (18, Figure 31) for temporary storage. The dewatering to a dry matter concentration of 25% is carried out with centrifuges (19, Figure 31). The dewatered sludge is further treated with green waste in the composting plant on the WWTP's area (21, Figure 31). The sludge dewatering effluent (SDE) is directed to the SDE treatment tank, treated via nitrification/nitritation processes and finally returned to the reaction tanks of the 1st stage after mixing in the collector (4, Figure 30) with the mechanically pre-treated wastewater and the returned activated sludge of the 1st stage.

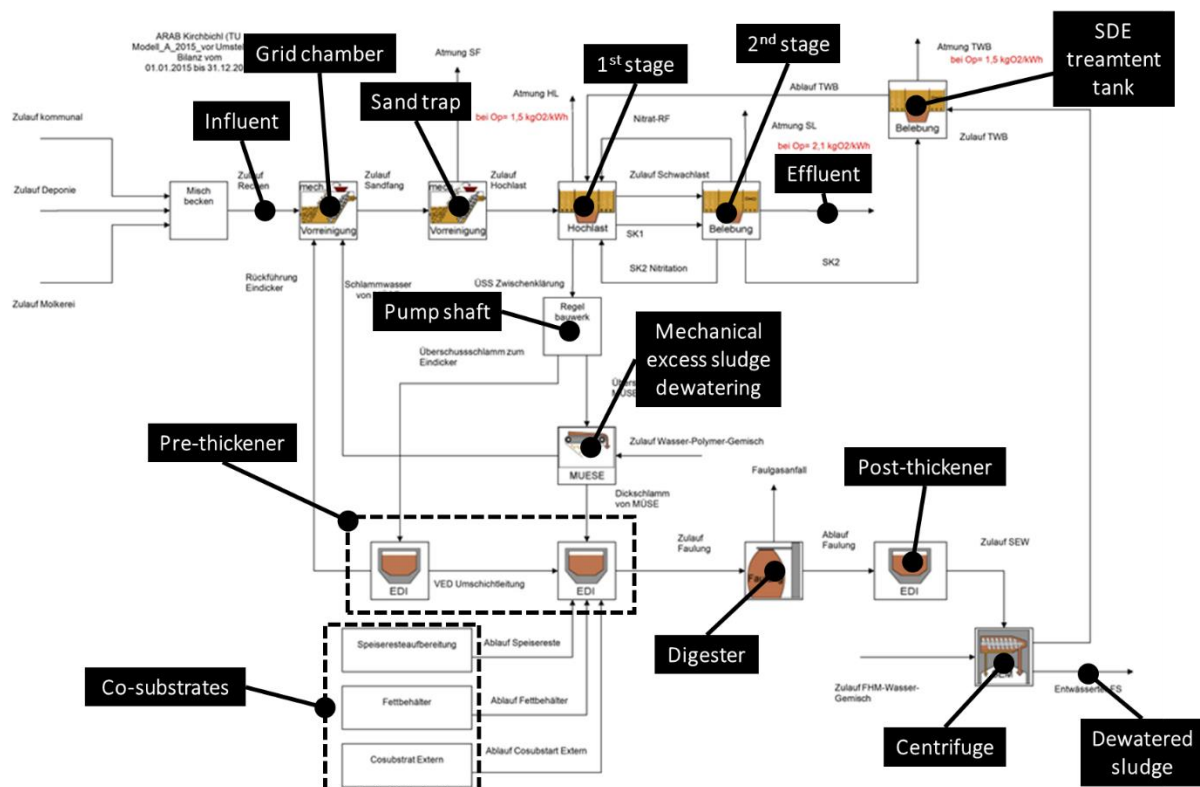


Figure 32: Flow scheme of the eDAB model

In accordance to the WWTP configuration and the sampling points a balancing model was developed with the software eDAB. The inflow of the total balance consists of the incoming wastewater via the sewer system as well as of the delivered co-substrates



(mainly household food wastes). Three output loads were defined: the discharged treated wastewater (WWTP effluent), the dewatered sludge as well as the off-gas from the aerated tanks and the digesters. For the period prior the operational change the balance period was set at 1 calendar year (01.01.2015-31.12.2015) and for that one after the operational change a duration at 5 months (23.03.2016-31.08.2016). The flow scheme including the modules of the detailed balances is shown in Figure 32.

Due to the implementation of the model on the WWTP and more detailed analyses of the influent, which are planned for the future, a splitting in three sub streams (municipal wastewater, wastewater from the indirect discharger and leachate) was necessary, nevertheless this was not relevant for the balance results. Following the wastewater line of the WWTP, the subsequent modules "grid chamber" and "sand trap" were added, assuming that no biological processes occur there. The module "1st stage" covers both the high-loaded aeration tanks as well as the corresponding clarifiers. Therefore, the returned activated sludge of the 1st stage as an internal stream of the module was not taken into account. Important outgoing streams of the 1st stage are the excess sludge to the module "pump shaft" and the sludge flow SK 1, directed to the 2nd stage of the WWTP. Similar to this, the module "2nd stage" includes the pre-denitrification tank, the three carousel aerated tanks and the secondary clarifiers. The returned activated sludge of the 2nd stage as an internal flow of the module was not considered in the calculations, but the sludge stream SK 2, piped into the SDE treatment tank prior the operational change and into the 1st stage after it. For completeness, the recirculation of the WWTP's effluent into the 1st stage for higher N and COD removal is also depicted (Nitrat RF), even if this water flow is negligible.

The excess sludge is split into two streams using the module "pump shaft". One of them passes the mechanical dewatering station and is mixed with the co-substrates in the mixing chambers of the pre-thickener. The second stream is returned back to the grid chamber via chamber of the pre-thickener. After the operational change this flow was turned off and only a small amount of sludge was piped to prevent clogging.

The main amount of co-substrates consists of household food waste, but also the amount of other kind of substrates can separately be recorded by the model. The dewatered sludge is mixed with the co-substrates in the mixing chamber of the pre-thickener and piped into the digesters (module "digester"). The effluent of the both parallel digesters is temporary stored in the post-thickener, the single unit in the module "post-thickener". The module "centrifuges" combines the centrifuges of the dewatering station as well as the filter press, which is currently out of operation. Even if the composting plant is within the cadastral border of the WWTP's area, it is not considered in the balance model. Therefore, the dewatered sludge is an outgoing stream in the eDAB model.

The sludge dewatering effluent is directed to the SDE treatment tank; its effluent is returned to the 1st stage.



5.1.3. Mass balances for Q, COD, TN and TP

The total water flow balance for the year 2015 shows a 4 % higher effluent compared to the influent water flow, which is in the acceptable range of mass balance deviations. Due to the buffering effect of the WWTP's tanks the measurement of the effluent was more balanced and therefore considered to be more reliable. Since no data about the flowrate and chemical characteristics of co-substrates were available, the daily inflow of co-substrates was set at 25 m³/d (this value was confirmed by the balances for the TP, TN and COD for the module sludge digestion). Figure 33 depicts the detailed water flow balance. The highest deviation was observed for the module "pump shaft", probably caused by an unreliable flowrate measurement of the stream directed to the pre-thickener. Returning a part of the excess sludge back to the influent prior the aerated sand trap can promote biological processes in the latter, which are not estimable due to the lack of measurements. This stream was turned off for the next monitoring period as an operational optimisation measure at the WWTP.

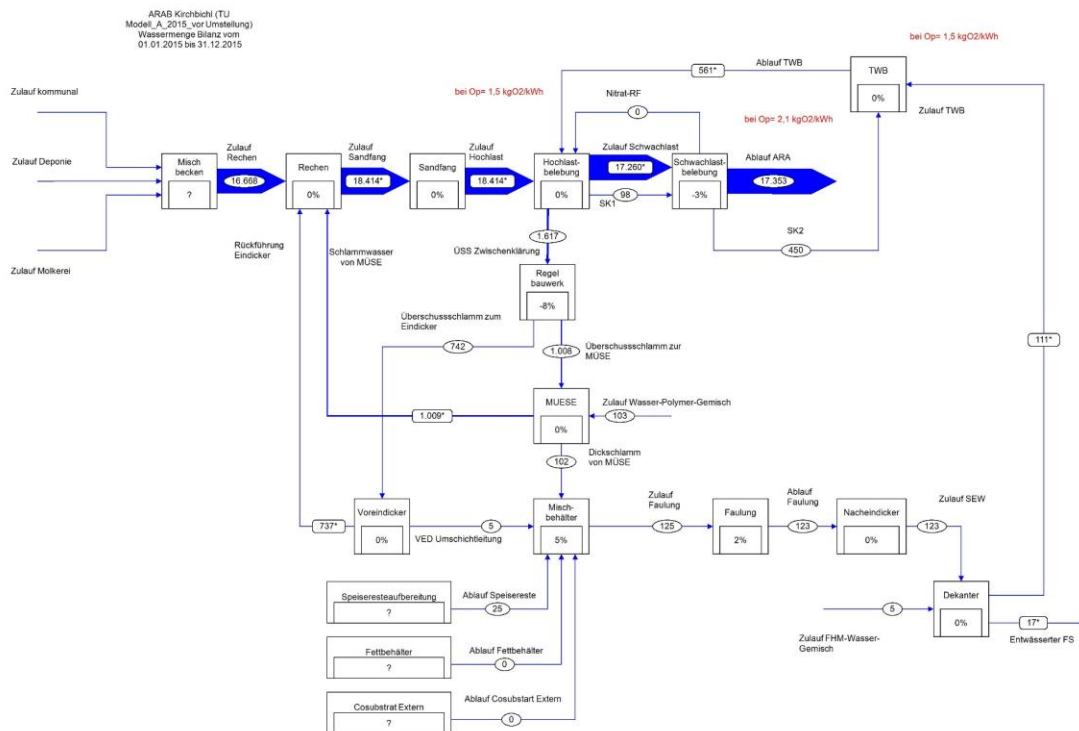


Figure 33: Detailed water flow balance of WWTP Kirchbichl for the period 01.01.2015-31.12.2015

The mass balances for MLSS are relevant only for mechanical treatment modules since the biological degradation of MLSS and the biomass growth in the aerated tanks was not quantified. In this regards, the high negative deviations for the modules 1st and 2nd stage result from the biomass growth in the reactor and its subsequently excess from the both biological stages. The balancing deviation of the module digester indicates the degradation of the volatile suspended solids which lies within the range of values reported in the literature. Nevertheless, MLSS loads were calculated for the internal streams and used for further estimations of the TP, TN and COD loads (Figure 34).



On WWTPs internal flows are usually not characterised by its chemical parameters, except in case of required monitoring. Prior changing the SDE treatment process, at the end of the balanced period 2015, samples were taken and analysed in the laboratory of the institute. The results for the ratios TP/VSS, TN/VSS and COD/VSS were comparable with results from the latter analysis program, so the data obtained were assumed to be representative and fit into the eDAB-model.

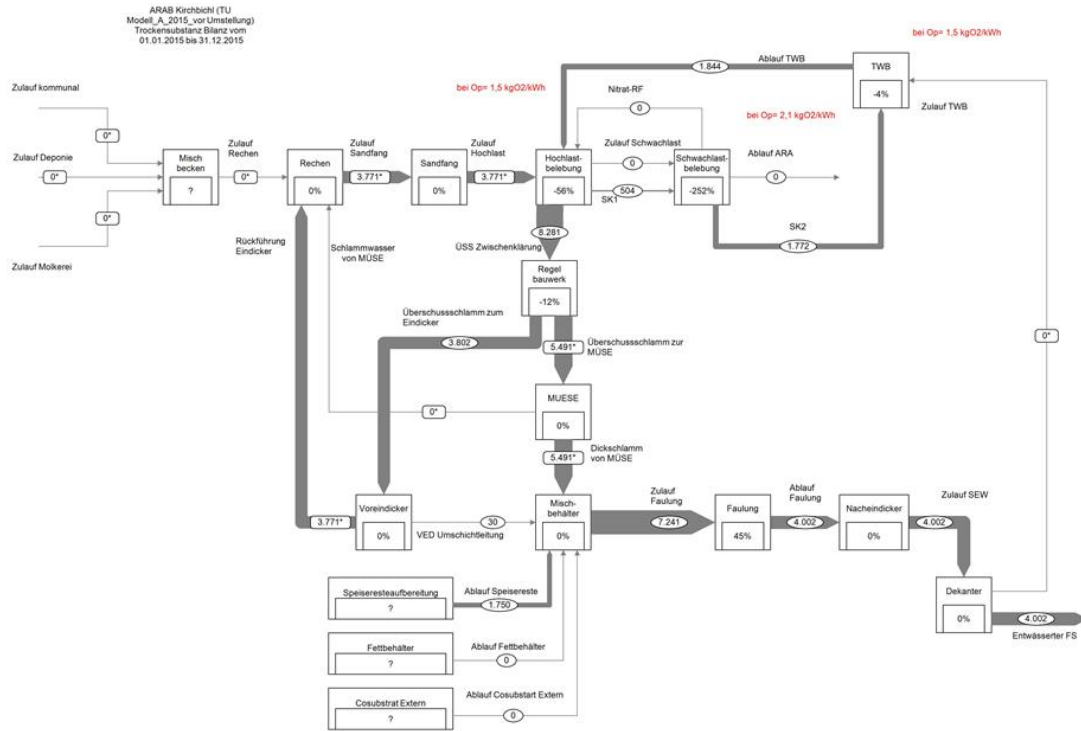


Figure 34: Detailed mass balance for MLSS of WWTP Kirchbichl for the period 01.01.2015-31.12.2015

A good quality for the TP mass balance was achieved, as can be seen in Figure 35. The deviation of the total mass balance was 7 % (output higher then input) and the phosphorus removal efficiency was 91 %.



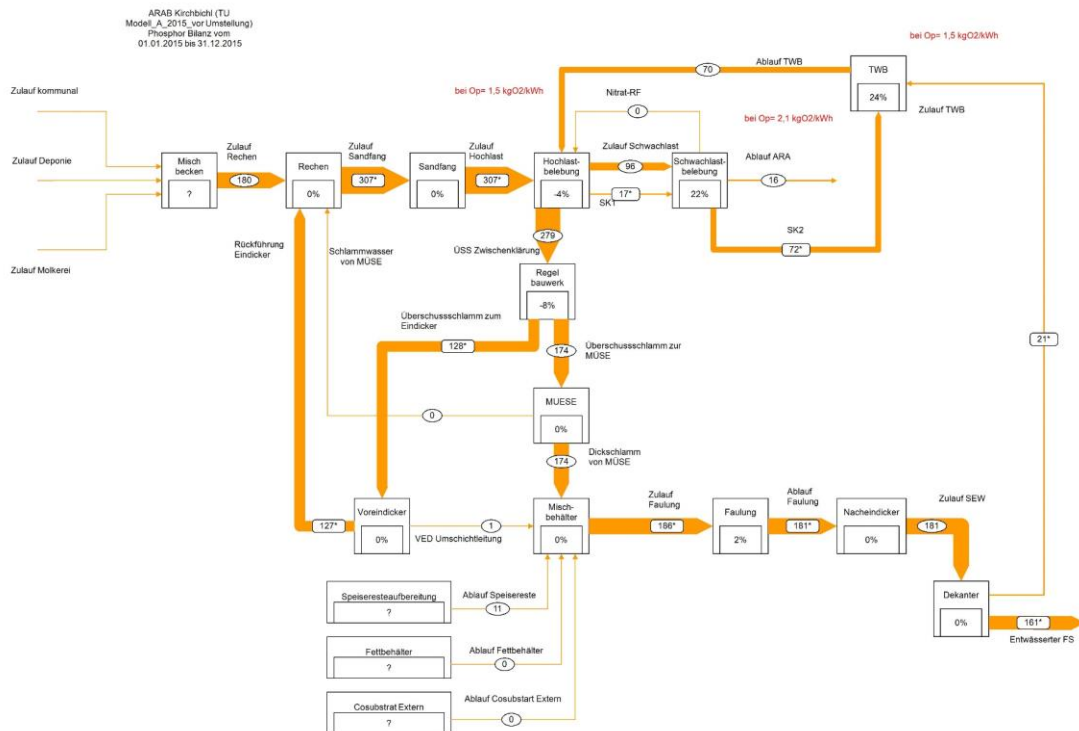


Figure 35: Detailed TP mass balance of WWTP Kirchbichl for the period 01.01.2015 31.12.2015

Higher gaps were observed for the modules 2nd stage and SDE treatment tank, which was led back to the WWTP data lack. The deviation for the module “pump shaft” results from the unreliable flowrate measurement of the stream to the pre-thickener and from the splitting of the TP-load of the excess sludge based on the flowrates ratio.

The detailed nitrogen mass balance is shown in Figure 36. A nitrogen removal efficiency of 67 % was calculated for the monitored period. The total nitrogen balance of the WWTP resulted in difference of 1 % (total load to the WWTP lower than the output loads). Since the nitrogen mass balance of the single modules (1st stage, 2nd stage and SDE treatment tank) is open due to the not quantifiable denitrified nitrogen gas, the gaseous loads are calculated as the difference of the single in- and output loads for each module. The main denitrification occurs in the 2nd stage of the WWTP (80 %), followed by the 1st stage (15 %) and the SDE treatment tank (5 %). The denitrification in the SDE treatment tank is limited by the low BOD in the SDE and in the returned sludge of the 2nd stage (SK 2).

On the contrary a closed balance for nitrogen is possible for the digester and resulted in a gap of 6 %. This value is within the tolerance range of ± 10 % and confirms the quality of the chemical analyses as well as the estimation of the mass load of the digester effluent.



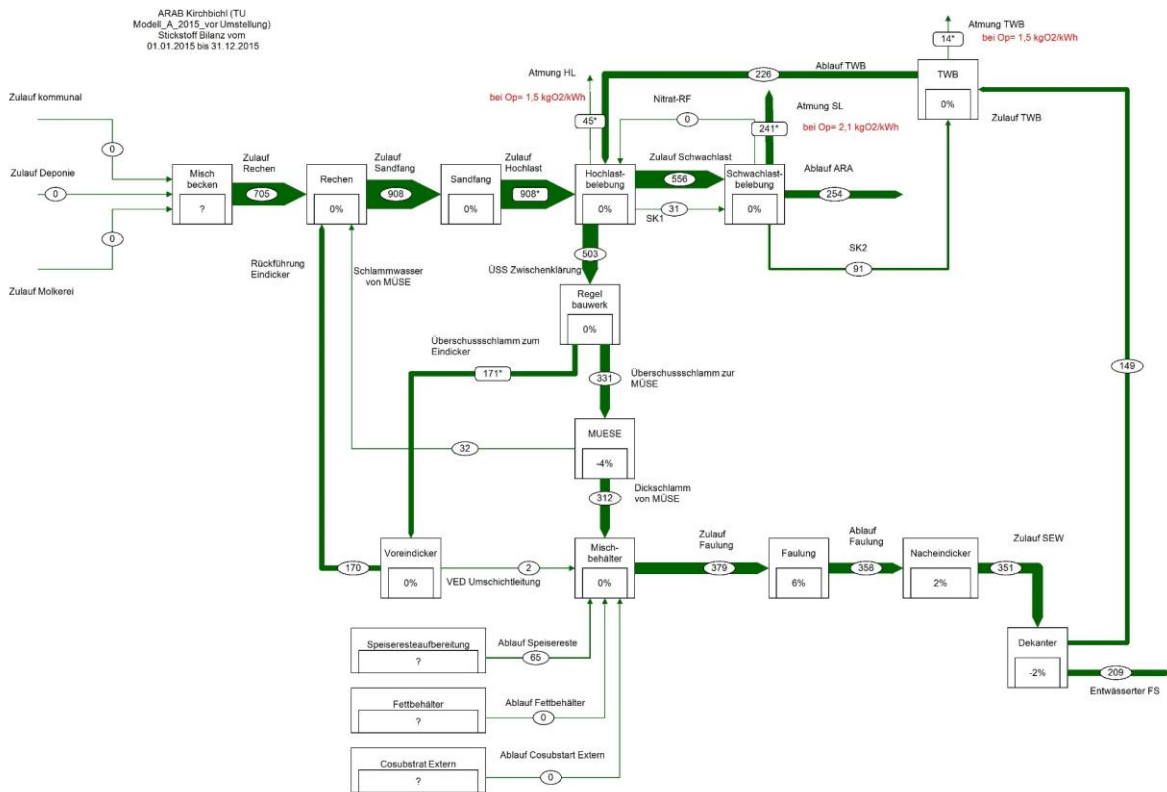


Figure 36: Detailed TN mass balance of WWTP Kirchbichl for the period 01.01.2015-31.12.2015

Regarding the COD total balance of the WWTP a 5 % higher input of COD compared to the sum of the output COD loads was calculated. The COD removal efficiency was 96 %. The main COD output is the produced biogas (50 % of the inflow COD), followed by the dewatered digested sludge (27 %) and COD load from the heterotrophs activity (14 %). The detailed COD balance is presented in Figure 37. The high quality of the balance for the module “digester” was confirmed by a gap of only 2 %. The main COD degradation for the wastewater treatment line occurs in the 2nd stage of the WWTP (88 %), followed by the SDE treatment tank (11 %) and the 1st stage (1 %).

The average N/COD ratio of the WWTP's influent was 0.078 and of the biological step influent was 0.067.



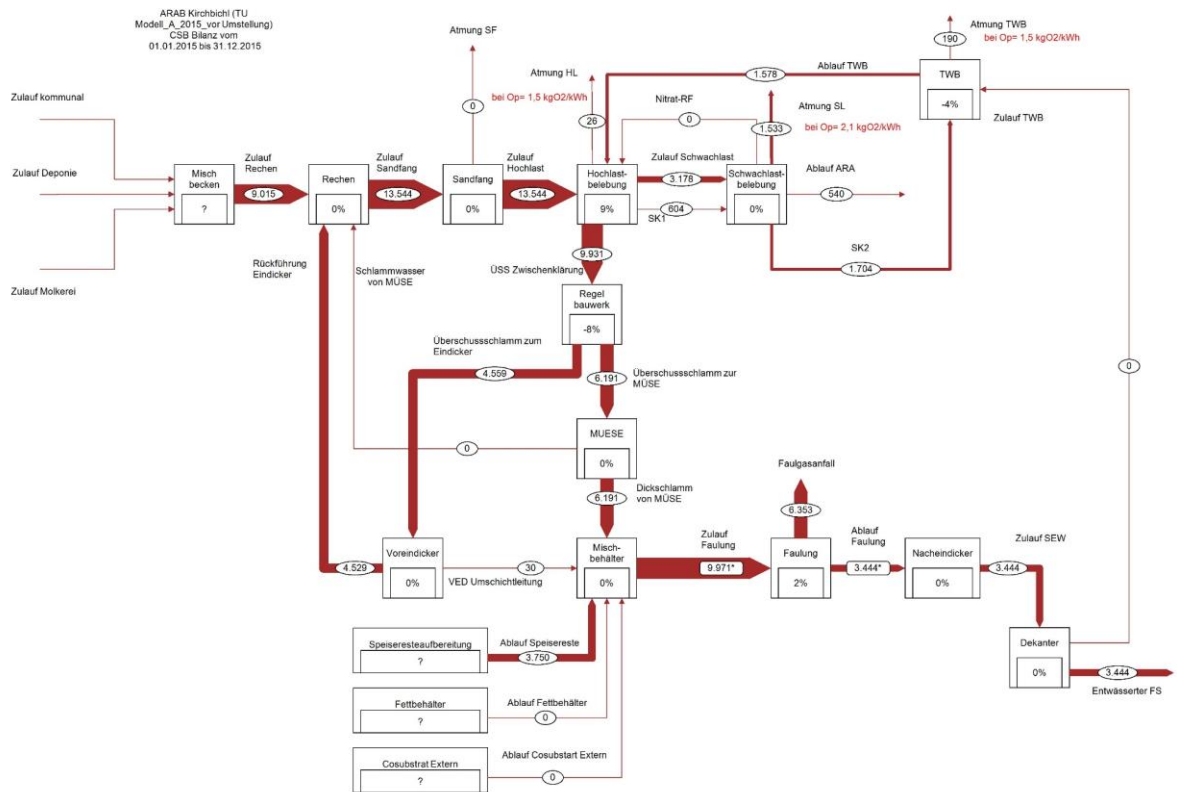


Figure 37: Detailed COD mass balance of WWTP Kirchbichl for the period 01.01.2015-31.12.2015



5.2. Treatment of SDE via Nitritation in side stream on the two-stage WWTP Kirchbichl

Prior to the change of the SDE treatment via nitritation, the SDE treatment tank was operated as a nitrification/denitrification process. The effluent from sludge dewatering was dosed to the SDE treatment tank. To supply nitrifying biomass, a partial flow of the return sludge from the 2nd stage (SK 2) was also returned into the SDE treatment. The amount of recirculated sludge (SK 2) was four times higher than the centrate from sludge dewatering.

The main consequence of the nitritation implementation process is the raise of ammonia concentration in the SDE treatment tank. This leads to an inhibition of the nitrite oxidizer (NOB) with the consequence of decreasing nitrate concentration and increasing amount of nitrite. To trigger the inhibition of NOB, it is necessary to prevent the dilution of the SDE effluent in order to ensure higher ammonia concentration in the treatment tank. Therefore, the recirculated sludge (SK 2) is bypassed and dosed to the 1st stage.

The operational parameters in the mesophilic digestion lead to a SDE with both, high temperature and ammonia concentration. The missing of dilution due to bypassed SK 2 leads to changing conditions in the SDE treatment tank with rising temperature and ammonia concentration which implicates an adaption of the microbial biomass.

5.2.1. Flow scheme of the modified operation

Focusing on the project aim two main operational changes of the treatment process, presented in Figure 38, were carried out on the WWTP Kirchbichl.

Since a stream of excess sludge from the 1st stage was returned back to the grit of the mechanical step, nitrification was assumed to occur also in the aerated sand trap, which makes the balancing of the 1st stage difficult due to lack of chemical analyses for the influent of the biological step. This flow was turned off, whereby only a small amount of excess sludge was passed through the pipes to prevent clogging.

To implement a nitritation for the treatment of the SDE while keeping the configuration of 2-stage Hybrid[®] WWTP, the excess sludge from the 2nd stage was redirected into the 1st stage (SK 2 Nitritation). These changes were finalised by the mid of March, so the new balance period covers a period of five months (23.03.2016 to 01.09.2016).



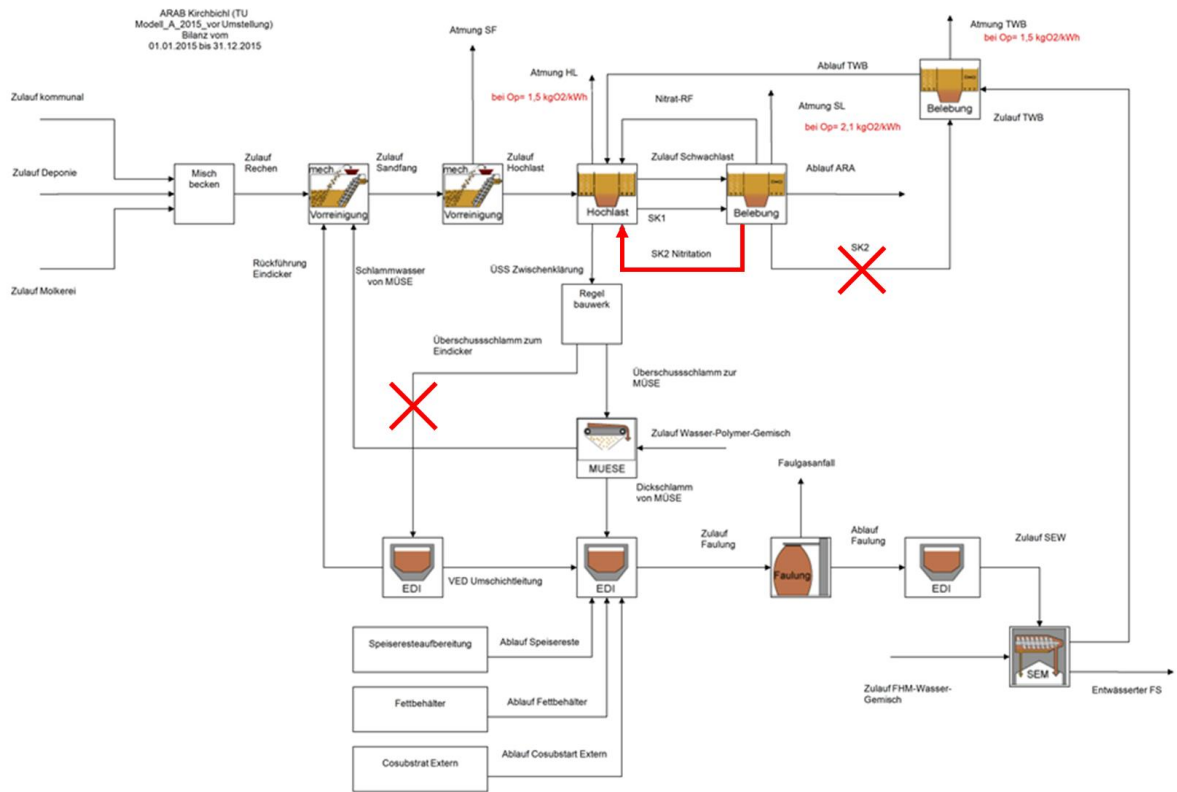


Figure 38: Modified flow scheme



5.2.2. From side stream nitrification to side stream nitritation

Changing the SDE treatment process from nitrification to nitritation is accompanied in some cases with the occurrence of foam formation. Foaming may be caused by a number of factors. The acclimatization of the microbial community is one common reason for such effect. Another reason for foaming may be the adjustment of the flocculant for sludge dewatering. During the start-up phase of the nitritation with redirection of the SK 2 from the SDE treatment tank to the 1st stage different polymers (liquids and powders) were tested. A possible overdosage of flocculants and the lack of dilution due to the bypassing of SK 2 may cause the formation of foam during the aerated phase in the SDE treatment tank. The following pictures (Figure 39) show the problems with occurrence of foam.



Figure 39: Foam formation in the SDE treatment tank Kirchbichl

Due to this problems, the start-up phase of the nitritation was investigated also in laboratory scale. The experimental reactors R1 and R2 were filled with nitrifying sludge from the 2nd stage. R1 and R2 were treated differently regarding the temperature in the liquid. R1 runs at room temperature (about 22 °C) and R2 runs at 35 °C. Boundary conditions like pH-value during feeding, O₂-concentrations (0.05 – 2.0 mg/L) in the reactors and ammonium concentration in the feeding SDE (1 600 mgNH₄-N/L) were similar.

The feeding pumps of both reactors were controlled by the pH-value in a range of 7.0 – 7.2. When the pH-value is lower than 7.0 the pump gets the signal to start charging SDE to the reactor. By dosing SDE to the reactor the pH-value rises. After reaching the upper limit (pH-value 7.2) the feeding pump gets the signal to stop. Due to acid production and consumption of acid buffer capacity during nitritation the pH-value declines. The feeding process starts again when reaching a pH-value lower than 7.0. Figure 40 shows the process of dosing SDE to the reactor.

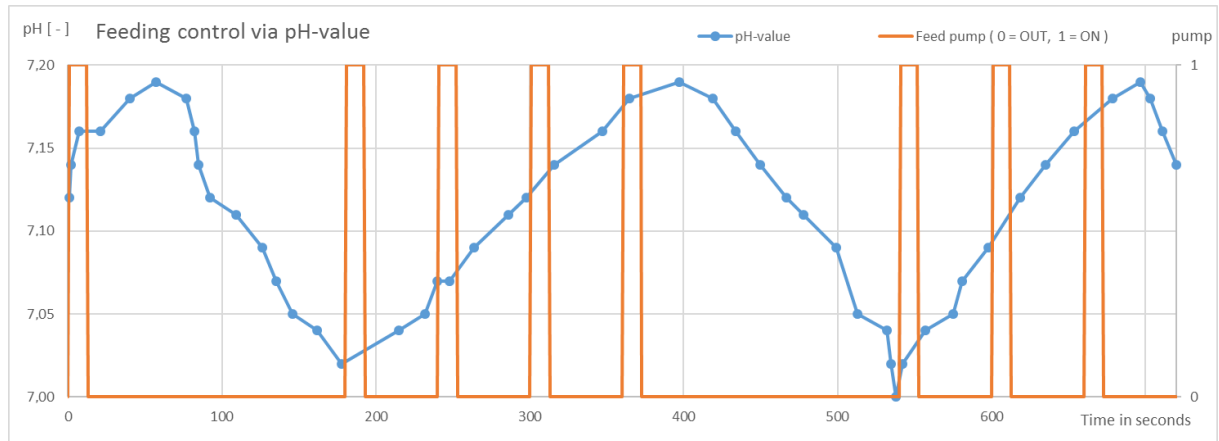


Figure 40: Feeding control of SDE via pH-value in laboratory scale

With these boundary conditions it can be ensured, that the conditions in the reactor are optimal for nitrificants. Lower pH-values may cause a delayed conversion of ammonium to nitrite or nitrate, pH-values much higher than 7.2 would raise the ammonia concentrations in the liquid with the consequence of beginning inhibition of both, AOB and NOB.

Experimental reactor 1 was started at room temperature of about 22 °C. The reactor contained 3 liters of sludge from the 2nd stage and was equipped with pH-sensor, oxygen-sensor, stirring unit and with an aeration system. The SDE was treated in a continuous flow process, which means that the hydraulic retention time (HRT) and the sludge age were equal.

The following Figure 41 shows the pH-value and the hydraulic retention time in R1. The loading rate of SDE results from the control of the pH-range (7.0 and 7.2) and the nitrification/nitritation rate.

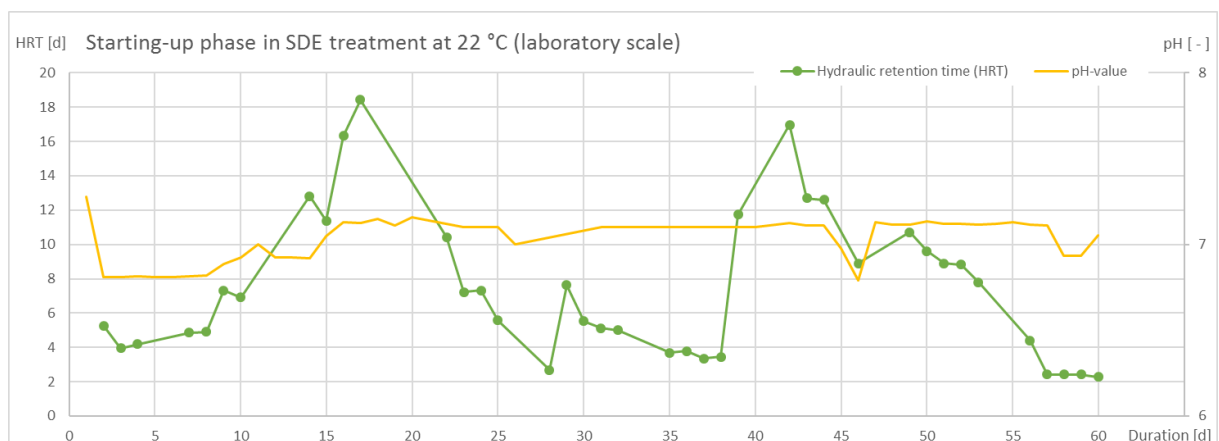


Figure 41: Start-up phase at laboratory scale at 22 °C

Figure 42 shows the SDE treatment effluent. At the beginning the dosed SDE is converted from $\text{NH}_4\text{-N}$ to $\text{NO}_3\text{-N}$. After 10 days rising $\text{NO}_2\text{-N}$ concentration in the effluent and decreasing $\text{NO}_3\text{-N}$ concentration could be observed, revealing the inhibition of NOB. A stable nitritation (conversion of $\text{NH}_4\text{-N}$ to $\text{NO}_2\text{-N}$) can be observed within 40 days after the start up.



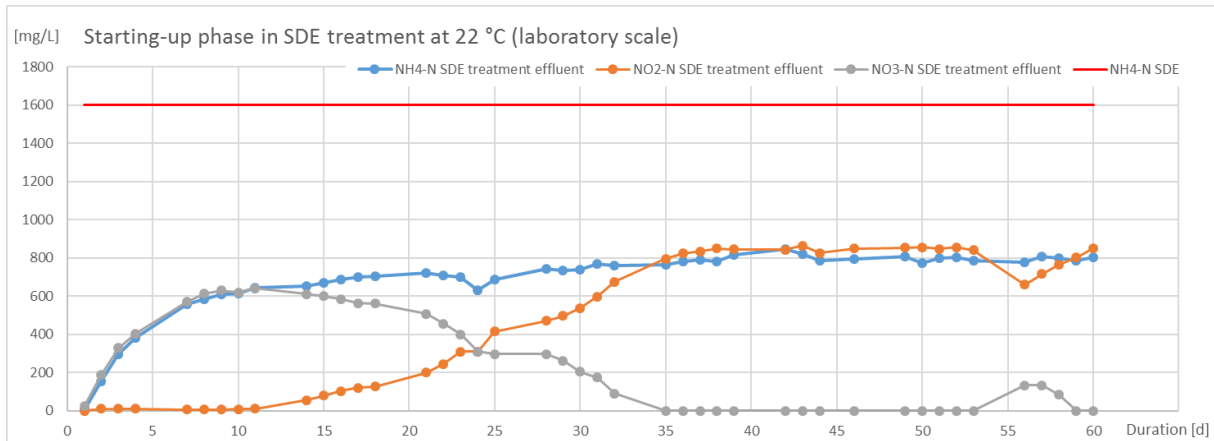


Figure 42: Performance of SDE treatment at laboratory scale at 22 °C

To investigate the influence of higher temperature, experimental reactor 2 was started at 35 °C. Reactor 2 is technically identical to reactor 1, contains 3 liters of sludge from the 2nd stage and is additionally equipped with a heating element. Similar to reactor 1, the pH-value (7.0 and 7.2) controlled the automatic feeding of SDE. The following Figure 43 shows the daily mean value of pH and hydraulic retention time (HRT).

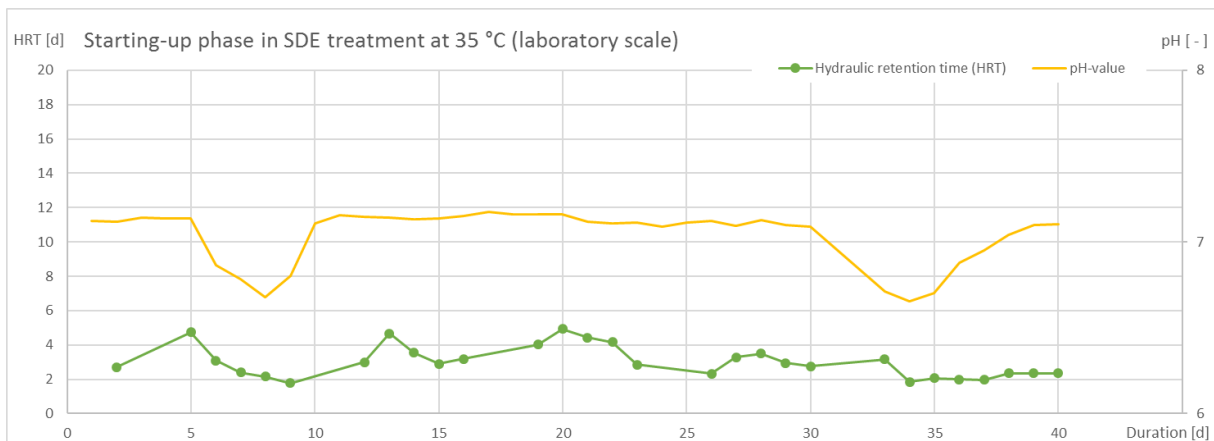


Figure 43: Start-up phase at laboratory scale at 35 °C

The diagram with the performance of reactor 2 (Figure 44) shows after only 5 days the beginning inhibition of NOB with the consequence of increasing NO₂-N concentration. A stable process of nitritation with conversion of approx. 50 % of the incoming NH₄-N to NO₂-N without the formation of nitrate can be observed after 15 days. The lower hydraulic retention time shown in Figure 43 also means a higher SDE loading rate to the reactor due to a higher conversion rate at 35 °C.



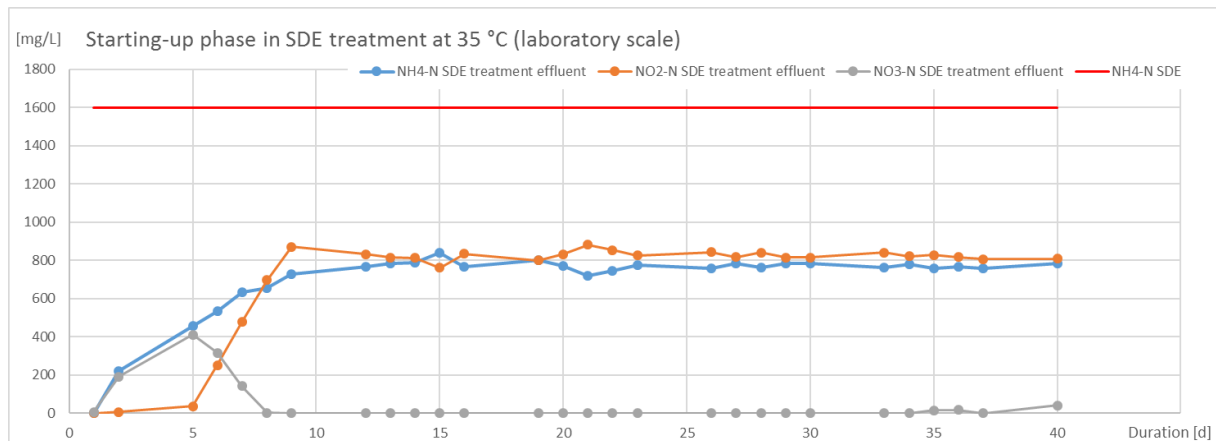


Figure 44: Performance of SDE treatment at laboratory scale at 35 °C

The SDE in Kirchbichl was previously treated via nitrification. On 21st of March the sludge from 2nd stage (SK 2) was redirected from the SDE treatment tank to the 1st stage to promote nitrification in SDE. Figure 45 shows the temperature and the pH-value in the SDE treatment tank as daily mean values. The temperature in the digesters is 39 to 40 °C. As a consequence, the temperature in the sludge dewatering effluent is correspondingly high. Due to the lack of dilution with SK 2 in the nitrification process, the temperature at the SDE treatment tank rose and reached a temperature of 30 °C. The pH-value lay between 6.5 and 7.0 most of the time.

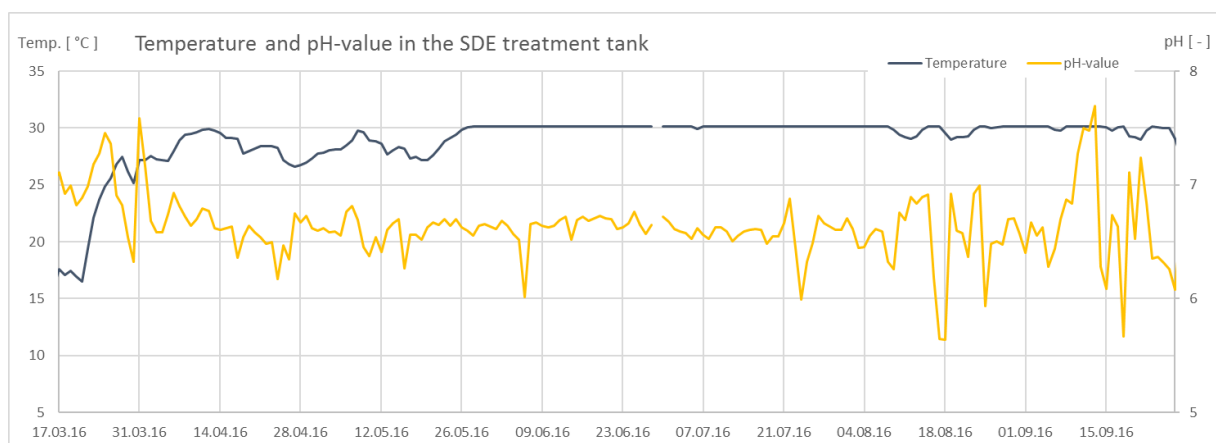


Figure 45: Temperature and pH-value in the SDE treatment tank Kirchbichl

Representative sampling from the beginning was hampered by the described foaming problems in the SDE treatment tank. After changing the polymer and finding the correct dose of the flocculant for sludge dewatering the foam formation decreased significantly. The first representative sample was taken on the 9th of May, 50 days after bypassing the SK 2 to the 1st stage of the WWTP Kirchbichl. The analyses of the SDE treatment effluent showed very low nitrate concentration and high concentrations of both, ammonium and nitrite (NH₄-N: 672 mg/L, NO₂-N: 670 mg/L, NO₃-N: 5 mg/L). The following Figure 46 illustrates the performance of the SDE treatment tank in Kirchbichl from day 50 to day 190. A stable nitrification without accumulation of nitrate can be observed over the whole period.



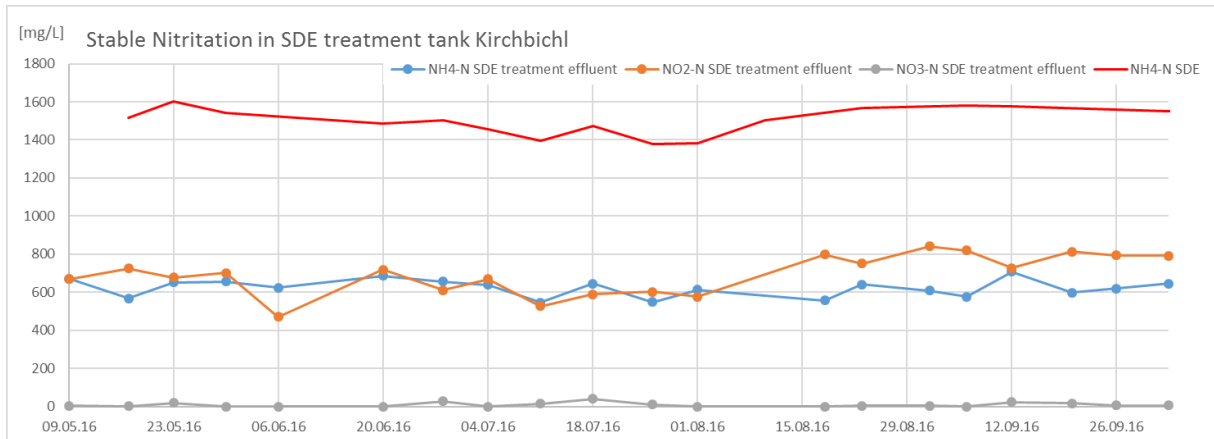


Figure 46: Stable nitritation in SDE treatment tank Kirchbichl (days 50 - 190)

The results of the experimental reactors R1 and R2, as well as the results of the SDE treatment tank in Kirchbichl are summarized in the following table (Table 15). The conversion of $\text{NH}_4\text{-N}$ to $\text{NO}_x\text{-N}$ (mainly $\text{NO}_2\text{-N}$) in the three types of SDE treatment tanks is in a narrow range between 51.3 and 52.9 %.

Table 15: SDE treatment - first results from laboratory-scale and full-scale in Kirchbichl

	Experimental reactor 1	Experimental reactor 2	SDE treatment Kirchbichl
Temperature	22 °C	35 °C	\bar{x} = 29.4 °C (25 °C – 30 °C)
Stable nitritation after	~ 40 d	~ 15 d	< 50 d
pH-value	7.0 – 7.2	7.0 – 7.2	\bar{x} = 6.6
O ₂ -concentration	0.05 – 2.0 mg/L	0.05 – 2.0 mg/L	0 – 1.5 mg/L
HRT	\bar{x} = 7.4 d (2.3 d – 18.4 d)	\bar{x} = 3.0 d (1.9 d – 4.7 d)	\bar{x} = 4.8 d
NH ₄ -N SDE	\bar{x} = 1 600 mg/L	\bar{x} = 1 600 mg/L	\bar{x} = 1 500 mg/L
NH ₄ -N treatment effluent	\bar{x} = 798 mg/L (days 40-60)	\bar{x} = 772 mg/L (days 15-40)	\bar{x} = 622 mg/L (days 50-190)
NO ₂ -N treatment effluent	\bar{x} = 816 mg/L (days 40-60)	\bar{x} = 824 mg/L (days 15-40)	\bar{x} = 689 mg/L (days 50-190)
NO ₃ -N treatment effluent	\bar{x} = 25 mg/L (days 40-60)	\bar{x} = 4 mg/L (days 15-40)	\bar{x} = 9 mg/L (days 50-190)
Conversion (NH ₄ -N → NO _x -N)	\bar{x} = 51.3 %	\bar{x} = 51.8 %	\bar{x} = 52.9 %

5.2.3. Adaptation of the WWTP sampling strategy for modified SDE treatment

The sampling points and measurements for the modified operation are presented in Figure 47. The need of a new sampling strategy arose from the lack of chemical analyses of the internal flows hampering the mass balances of several WWTP's modules as described in chapter 5.1.3.

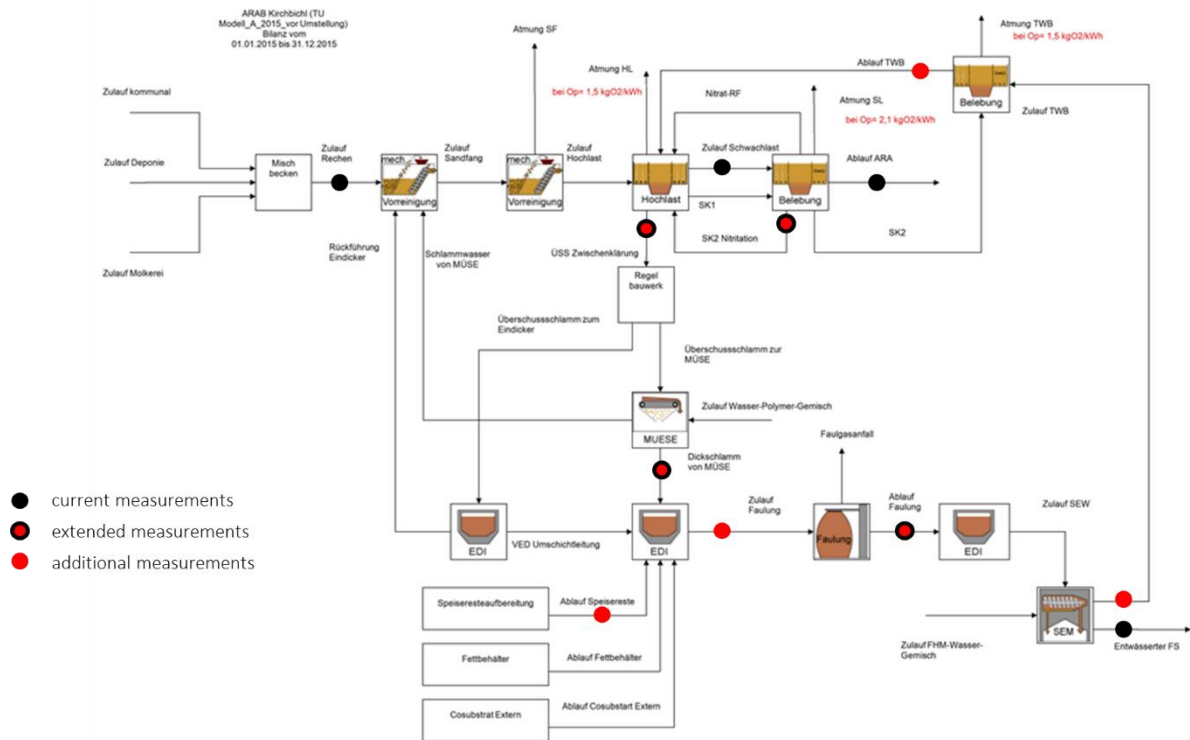


Figure 47: Flow scheme of the WWTP Kirchbichtl with sampling points for the modified operation

The new sampling plan of the WWTP was drawn comparing the existing sampling with the set of data necessary for balancing. Both are presented in Figure 48. Since analyses of the influent and effluent of the WWTP are required by authorities and necessary for the calculation of the removal efficiency, both streams are regularly analysed on the WWTP. Data of the effluent of the 1st stage (effluent of the clarifier) are also available. As shown in the table, the analyses for all internal flows sampled comprise only data for MLSS and VSS, which is an insufficient data set for detailed mass balances.

As mentioned in Chapter 4.1.2 the calculation of the substance loads is particularly based on the MLSS load and ratios Pp/VSS, Np/VSS or CODp/VSS. Therefore, further analyses for total parameters are required for the successful estimation of the loads and balancing of the investigated modules. Of high importance are the defined parameters for excess sludge from the 1st and 2nd stage as well as for digested sludge since the latter is a main output stream of the total balances as well as of the closed mass balances of the digester.

Focused on the changed SDE treatment process, the monitoring of the quality of the influent and the effluent of the SDE treatment tank is important due to the new concept for O₂ supply in the 1st stage via nitrite. Therefore, measurements of the dissolved nitrogen compounds are essential.



	COD	BOD ₅	TN	NH ₄ -N	NO ₃ -N	NO ₂ -N	TP	PO ₄ -P	MLSS	VSS
Influent	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
Effluent 1 st stage	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
RAS 1 st stage	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
RAS 2 nd stage	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
thickened sludge	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
gravity thickened effluent	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
raw sludge	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
digested sludge	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
dewatered sludge	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
SDE influent	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
SDE effluent	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied
Effluent	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied	Applied

Figure 48: Comparison of the applied and the necessary sampling on WWTP Kirchbichl



5.2.4. Mass balances for Q, COD, TN and TP

The calculated average ratios for the weekly analysed grab samples are summarised in Table 16. The obtained values were stable for the monitored period of 5 months and corresponded well to typical empirical values for the sludge flows.

Table 16: Results of the chemical analyses for the ratios of selected sludge flows

	Np/VSS	Pp/VSS	CSBp/VSS
RAS 1 st stage	0.082	0.043	1.69
RAS 2 nd stage	0.089	0.071	1.39
Digested sludge	0.081	0.073	1.44

Due to the fact, that the biomass in the SDE treatment tank did not flocculate like the one in the activated sludge tanks of the biological step in main stream, but was much more dispersed, the measurement of the MLSS was not applicable. Both, a 0.45 µm filtration of the effluent as well as the determination of the dry residues would lead to overestimation of the results due to the high concentration of dissolved salts (approx. 1400 mg/l dissolved nitrogen). Therefore, the samples were not analysed for MLSS and VSS.

Based on the laboratory analyses and the calculated ratios of the TP, TN and COD, the WWTP was balanced for the period mid-March to September 2016 (5 months) under the modified SDE treatment process. The balances were used to investigate the effect of the SDE nitritation against the SDE nitrification process.



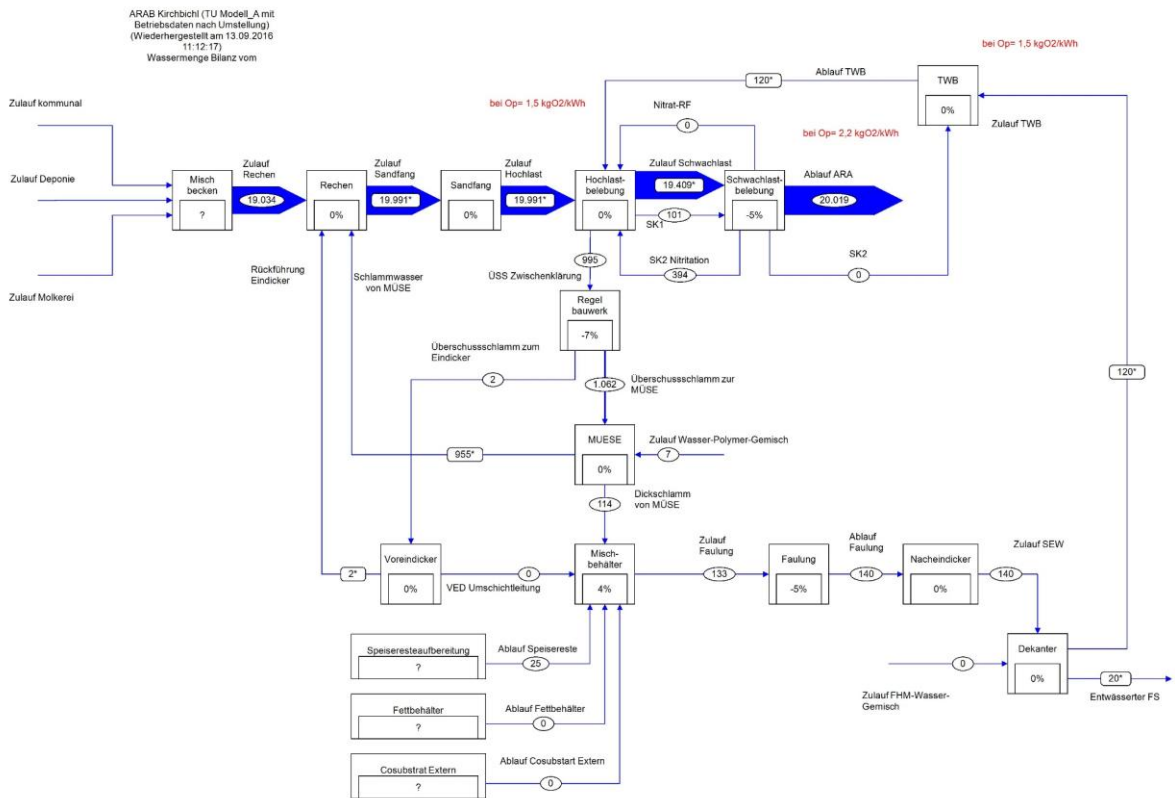


Figure 49: Detailed water flow balance of WWTP Kirchbichi for the period 23.03.2016-31.08.2016

The total flow balance for the monitored period resulted in a 5 % higher effluent flow compared to the influent flow and confirmed the results of the first monitoring period. A 10 % higher influent compared to 2015 was observed. The gaps for the single modules are comparable with the ones of the previous balance period.

Regarding the detailed mass balance of MLSS (Figure 50) comparable results were obtained for the balanced modules. Since the deviation for the post-thickener and sludge dewatering module ($\pm 5\%$) was within the acceptance range, no correction of the loads was undertaken. Strong analogy to the period prior operation change was objected for the module 1st stage. Even after the returned excess sludge stream back to the 1st stage was turned off, a huge amount of excess sludge is further removed from the 1st stage. Since the calculation of the stream is based on MLSS and flowrate measurements, additional MLSS measurements were undertaken to assure the data set. The average online measured MLSS concentration of the excess sludge was 8.2 g/l (5.1 g/l÷13.8 g/l), the MLSS concentration from the grab samples analysed at the institute's laboratory was 6.7 g/l (4.9 g/l÷9.2 g/l). Since online sensors shows often an off-set compared to grab sample concentrations and the measurement is based on predefined calibrations the results from the laboratory analyses were used for further calculations. Nevertheless, this high MLSS-load was verified through the module balance of the digester.



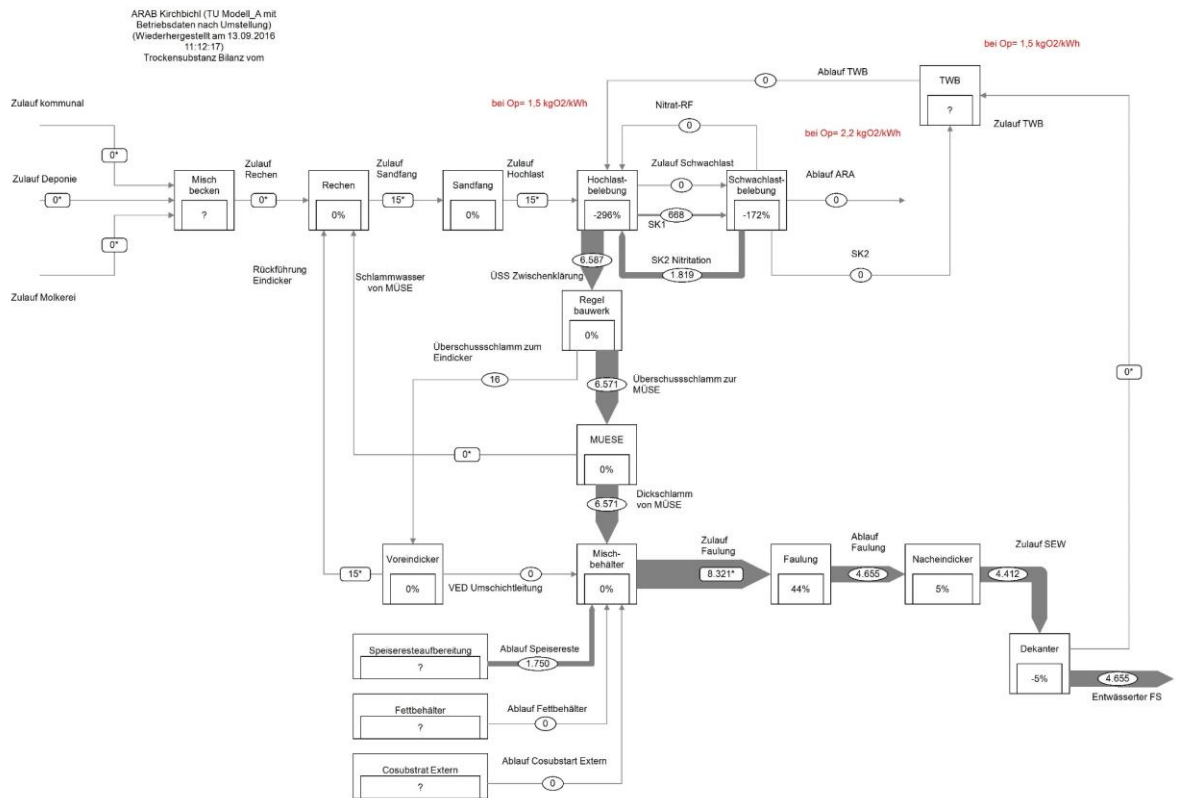


Figure 50: Detailed mass balance of WWTP Kirchbichl for the period 23.03.2016-31.08.2016

Regarding the TP balance a comparable removal efficiency of 92 % and a balance gap of -9 % was obtained. The 10 % higher inflow correlates to the higher water flow for the balanced period of 5 months. Higher deviations for the module 1st stage (-16 %) and digester (11 %) were observed. Due to the opposite trends the deviations were related to the higher estimated excess sludge load from the 1st stage. Taking into account, that a longer period increases the quality of the balance, it cannot be excluded that the shorter balance period of 5 months (compared to the previous period of 1 year) also plays a role for the higher observed gaps for all balances. It should be mentioned, that phosphorus mass balances in general show the highest gap.



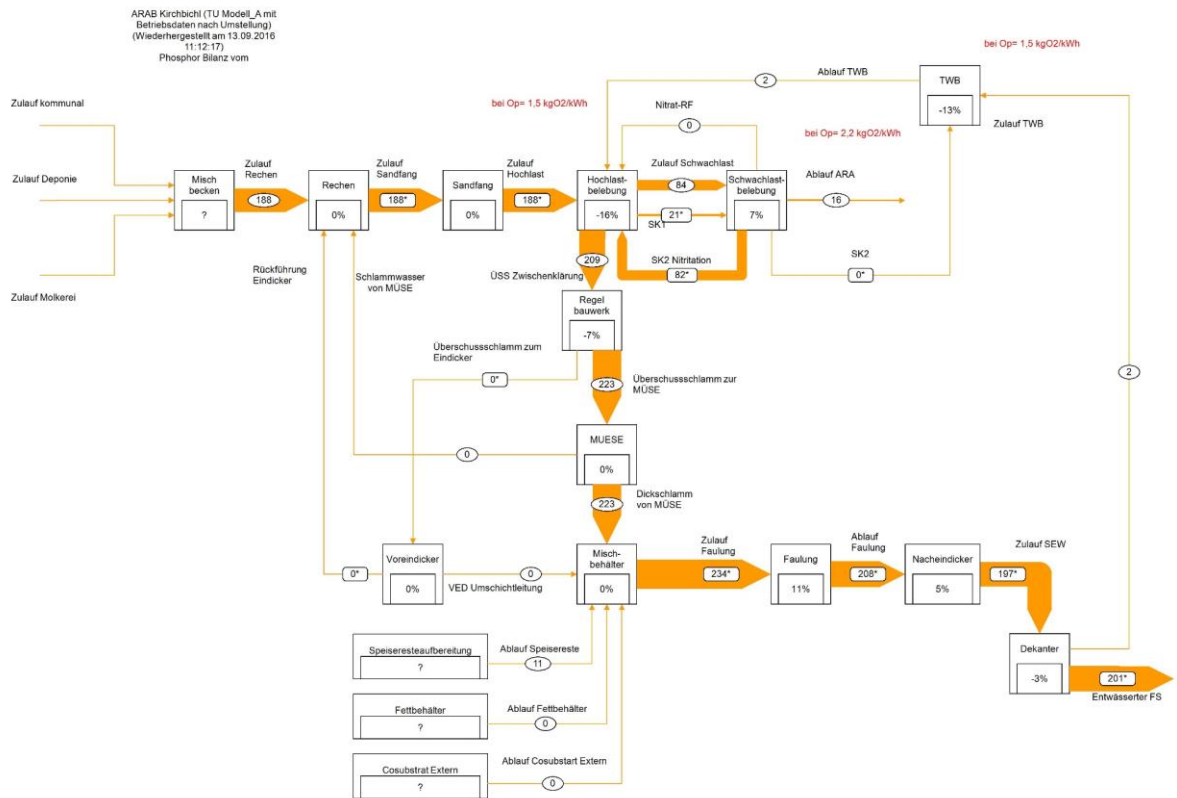


Figure 51: Detailed TP balance of the WWTP Kirchbichl for the period 23.03.2016-31.08.2016

Better results were obtained for the total nitrogen and COD mass balances. The total nitrogen mass balance gap for the entire WWTP was 4 %. The inflow was 8 % higher and was derived to the higher water flow for the monitored period. The determined average nitrogen removal efficiency dropped to 65 %. A good quality of the balances was obtained for the modules digester and sludge dewatering (Figure 52). Compared to the treatment process prior changing the operation a better denitrification in the 1st stage was achieved (27 % vers.15 % for the previous period 2015). The total nitrogen gas load for the entire WWTP amounted to 32 %, somewhat lower compared to the previous balanced period (39 %).

While the nitrogen load from the incoming wastewater increases by 10 %, the COD load remained stable and increased only by 2 %. Therefore, the average N/COD ratio of the influent increased to 0.084 and probably effected the nitrogen removal efficiency after the operational change. In contrast to the balance prior the modification, the gap of the total mass balance for COD was -5%, nevertheless a COD removal efficiency of 96 % was estimated. The COD removed by heterotrophs was slightly lower (1 579 kg COD/d) compared to the one prior changing the operation (1 749 kg COD/d). The higher biogas production after changing the operation can be mainly attributed to the higher COD load of the co-substrates.

Due to the redirected SK 2 no available biodegradable COD in the SDE treatment tank was assumed, so the COD degradation was set to zero. According to the balance 10 % of the COD was degraded in the 1st stage and 2nd stage remained the main contributor to the COD removal in the activated sludge tanks at the WWTP (90 %).



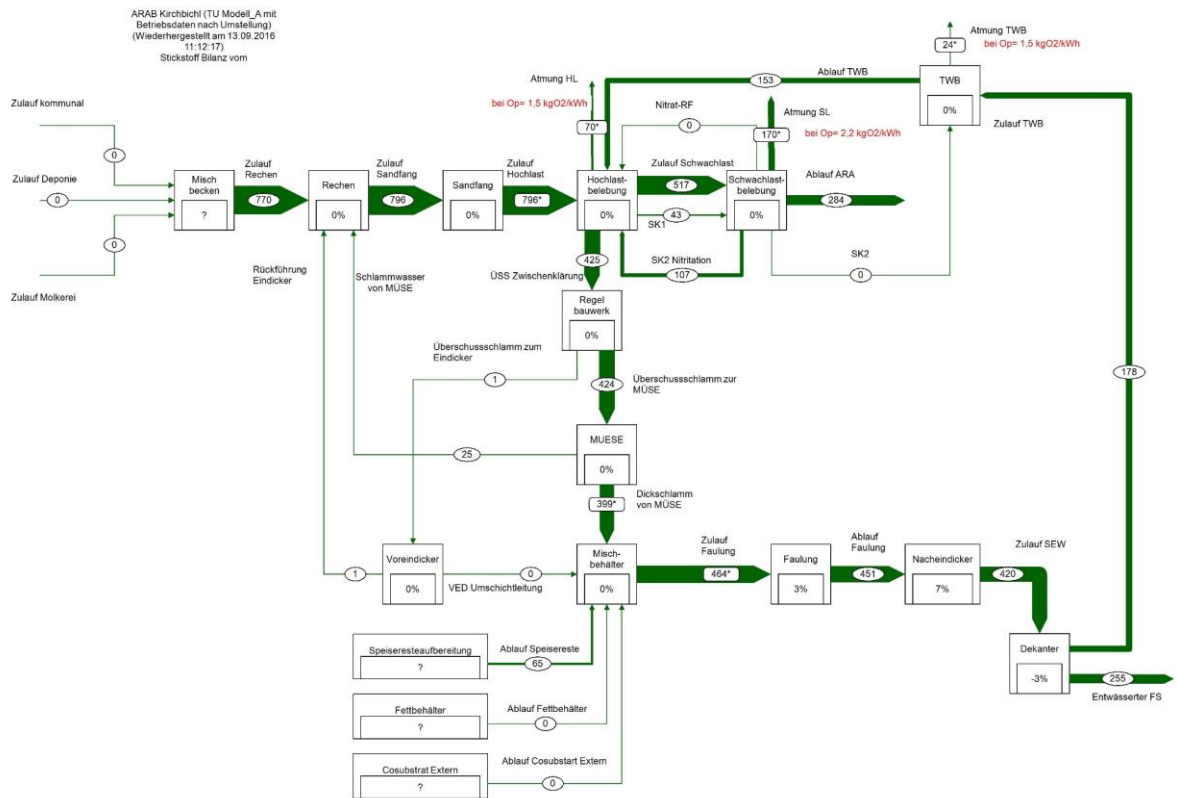


Figure 52: Detailed TN balance of the WWTP Kirchbichl for the period 23.03.2016-31.08.2016

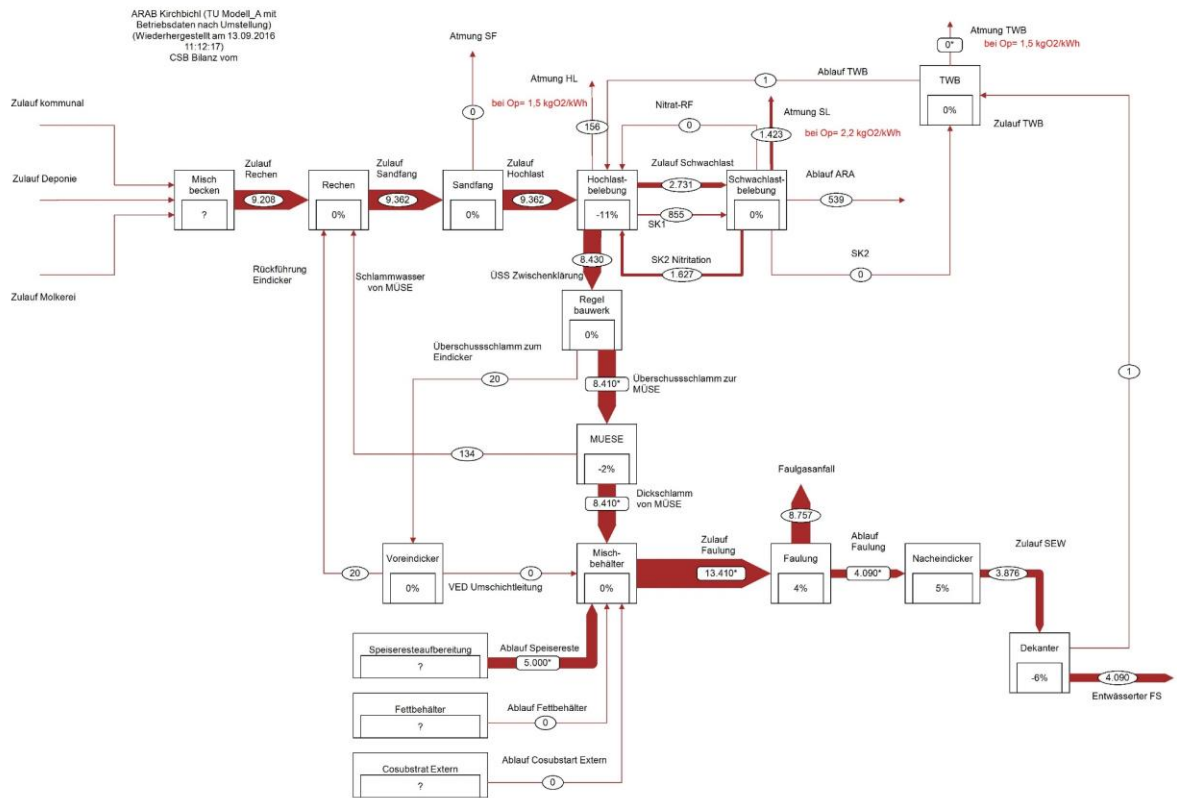


Figure 53: Detailed COD balance of the WWTP Kirchbichl for the period 23.03.2016-31.08.2016



5.3. N₂O measurements prior and after changing the operational conditions of the SDE treatment

The modification of the SDE treatment process was not accompanied by any changes of the feeding strategy of the SDE treatment. The SDE was pumped, as prior operational change, without intermediate storage direct into the treatment tank. Since under the new process conditions the SDE is not anymore diluted by the nitrogen low loaded sludge stream of the 2nd stage (SK 2), the NH₄ concentration in the SDE tank increased from approx. 200 mg NH₄-N/l (grab sample) to approx. 600 mg NH₄-N/l. As observed in previous measurements (Baumgartner and Svardal, 2014) the start-up of the nitrification process was accompanied by foam formation probably due to a shift in the biomass composition. Additionally, at the WWTP Kirchbichl the addition of the dewatering additives was not optimised at the time of the N₂O measurement campaign, so that a high amount of polymers is added to the digested sludge for dewatering. This, combined with the lack of dilution after the operational change of the SDE treatment tank, led to the intensive foaming (Figure 39). Since the N₂O measurement equipment is sensitive to humidity and high risk of foam suction exists due to the limited free space under the floating hood, the measurement campaign was postponed till the foam formation returned to normal.

During the stabilisation period of the SDE nitrification process, additionally to the proposed measurements on WWTP Kirchbichl, N₂O emission measurements were carried out at an another WWTP (WWP I), which is similarly designed as two-stage Hybrid[®] process. At WWPT I the SDE is treated as at Kirchbichl prior switching to nitritation (nitrification). The results are shown in Figure 54. The aim of this campaign was to observe the N₂O emissions for the same WWTP configuration under different influent characteristics.

The depicted measurements are already corrected for the N₂O concentration of ambient air, cross sensitivity of the measurement equipment to CO₂ as well as for the required dilution with ambient air to keep the N₂O concentrations within the range of the equipment.

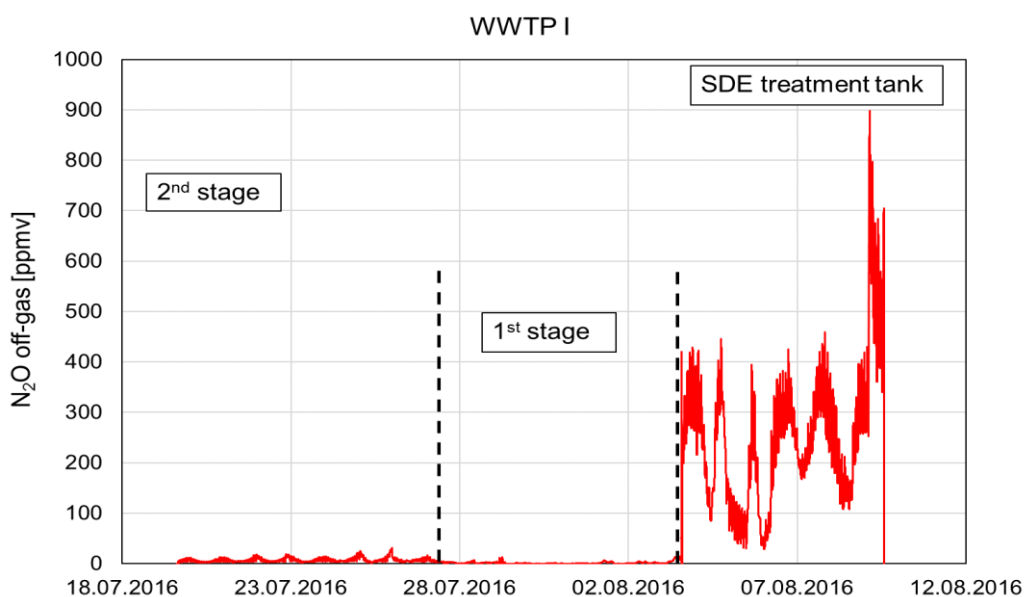


Figure 54: Graph of the N₂O off-gas concentration on WWTP I



The measurement started in the activated sludge tank of the 2nd stage of WWTP I for a period of one week (19.07-27.07.). The diurnal variation of the N₂O concentrations in the off-gas shows, that the highest values were reached at midnight and the minimal against 10 h (Figure 55), which corresponds to the daily maximal nitrogen load of the tanks. Due to the intermittent aeration of the 2nd stage, the N₂O concentration of the off-gas varied for each aeration interval. As soon as the aeration was turned on, a dilution of the head space of the floating hood with off-gas from the aeration tank occurs and the N₂O concentrations dropped, indicating that no N₂O accumulates during the denitrification. Due to stripping, the N₂O concentration started to increase and reached the maxima at the end of the aeration phase.

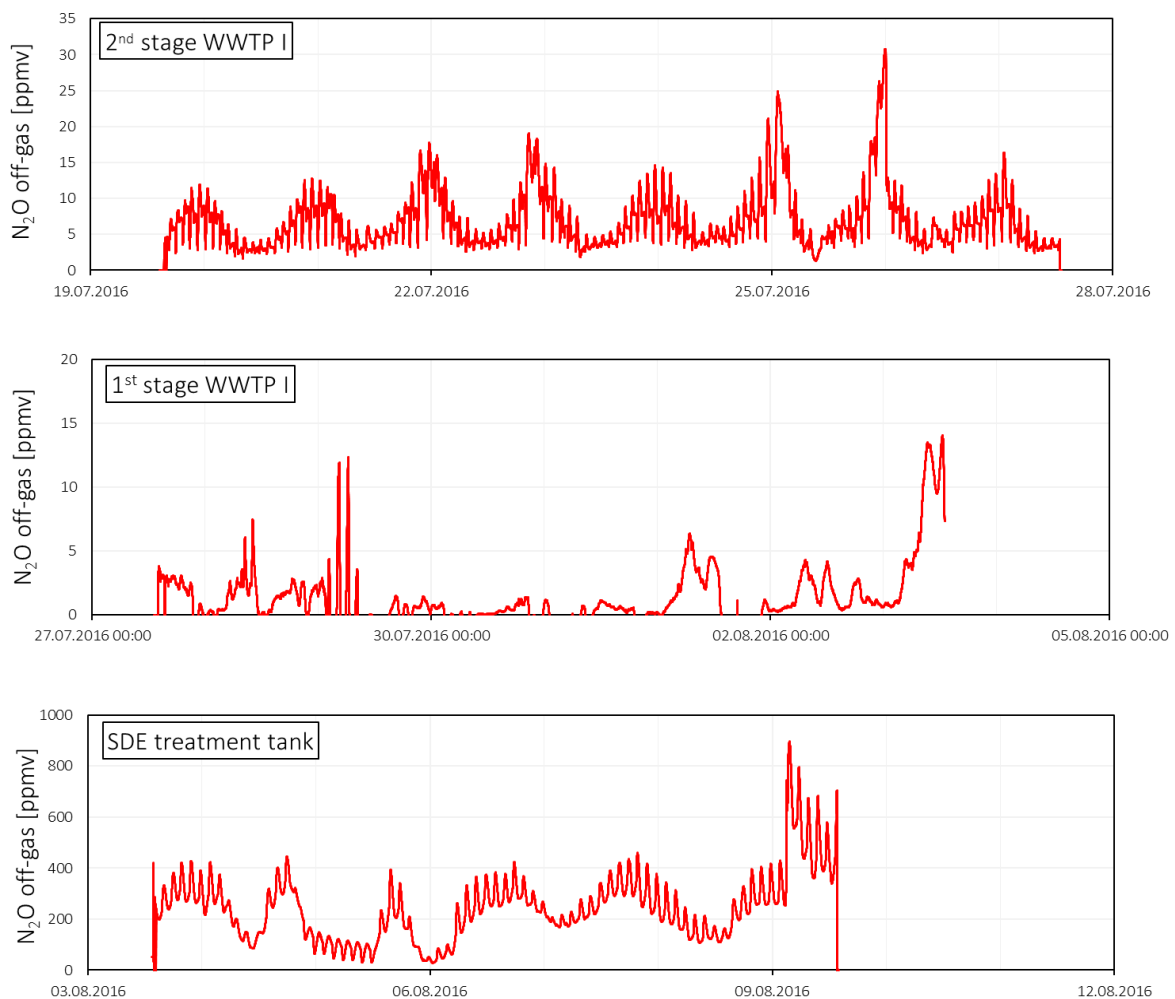


Figure 55: N₂O off-gas concentration measured at WWTP I

For the continuously aerated activated sludge tank of the 1st stage, the N₂O concentrations didn't follow any diurnal pattern. As the results show, the N₂O concentrations are in general much lower than the ones measured in the 2nd stage. The N₂O emission from this aerated tank is mainly related to stripped N₂O originating from the partial flow of treated SDE, returned to the 1st stage.

The SDE treatment at WWTP I is carried out via nitrification and the inflow streams are designed similarly as on WWTP Kirchbichl. The both SDE treatment tanks are intermittently loaded with SDE and excess sludge from the 2nd stage during the not aerated phase.



During the subsequent aeration, the treated water overflows the effluent weir and is discharged in the 1st stage. Since no preceding storage tank exists, the SDE treatment tank is loaded only during the sludge dewatering, which explains the diurnal variation of the N₂O concentration.

Once the nitrification of SDE was implemented and the foam formation returned to normal, the N₂O off-gas measurements were carried out also at WWTP Kirchbichl. The online monitoring period was 10.08.2016 to 01.09.2016. The results for the N₂O off-gas concentrations are presented in Figure 56. At Kirchbichl significant higher N₂O off-gas concentrations were measured not only for the 2nd stage, but also for the SDE treatment tank. Due to a revision of the centrifuges, the SDE treatment tank was not regularly loaded. Matching the observation at WWTP I the N₂O off-gas concentration from the SDE treatment tank was also at WWTP Kirchbichl the highest. Moreover, it exceeded by far the N₂O concentrations at WWTP I. As soon as the SDE treatment tank was regularly charged, the N₂O off-gas concentrations showed a typical variation related to the feeding plan. Since the 1st stage of the WWTP is rarely aerated and N₂O is emitted mainly during the aeration, the N₂O measurements were conducted here only for 18 h.

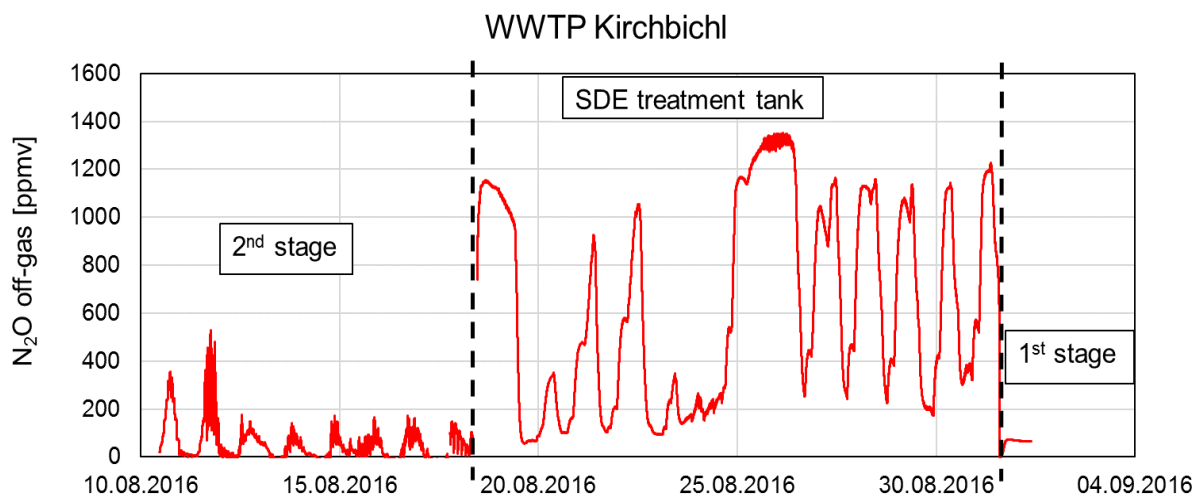


Figure 56: Graph of the N₂O off-gas concentration on WWTP Kirchbichl

The measurement was started in the 2nd stage of the WWTP. Due to rainfalls on 09.08. and 10.08. the inflow increased by the double of the dry weather inflow and caused significant higher N₂O concentrations (2- to 3-fold higher compared to the average daily maxima for dry weather). The diurnal variation, observed on WWTP I was confirmed by similar N₂O daily pattern (Figure 56).

As the measurements were carried out only in one of the parallel aeration tanks of the 2nd stage, for estimation of the whole N₂O emission from the 2nd stage a comparison and verification of the equality of the three tanks is required. As shown in Figure 57 exemplary for 1 day, the aeration phases and the oxygen concentrations in the three tanks are comparable. The higher NH₄-N concentrations in aeration tank 3 were led back to a higher calibration baseline of the corresponding NH₄-N sensor.

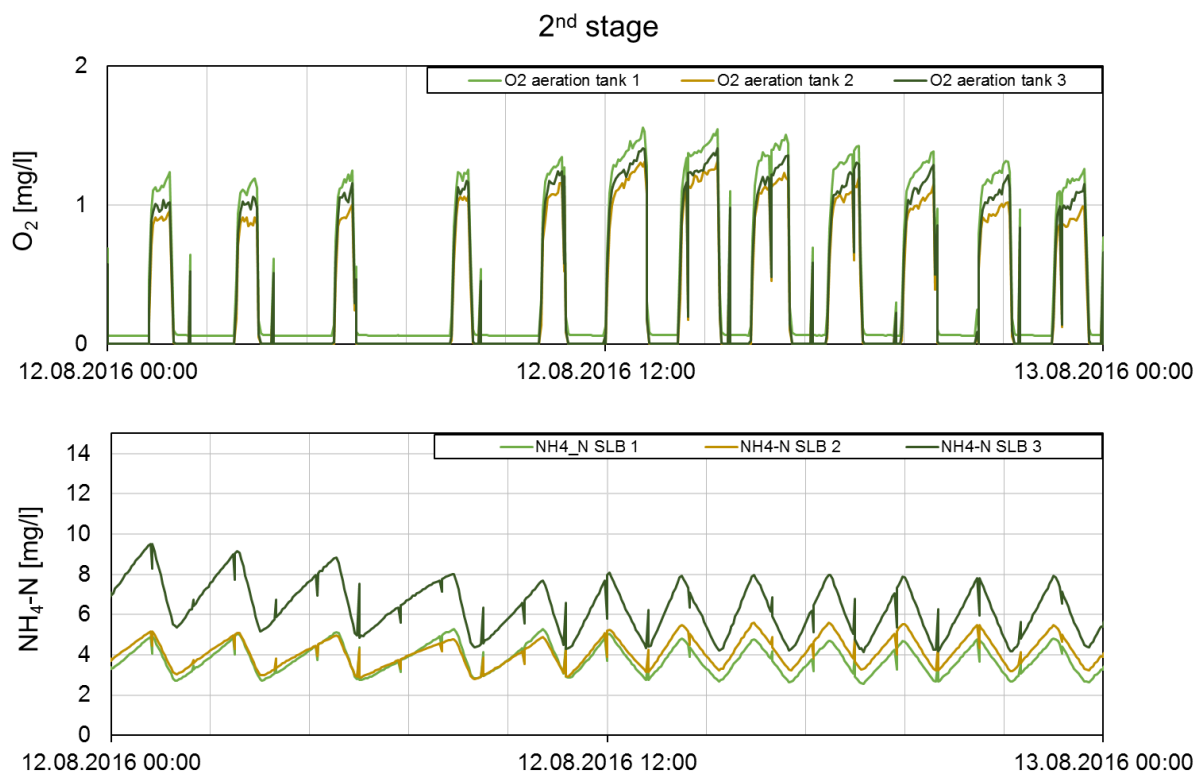


Figure 57: Comparison of the oxygen and ammonia concentrations of the three parallel aeration tanks of the 2nd stage of WWTP Kirchbichl

The average O₂ concentration of approx. 1-1.4 mg/l is slightly higher in aeration tank 1 due to a drift, detected during the anoxic phases. Since the oxygen supply in the 2nd stage is controlled by the ammonia concentration, intensive aeration with more often aeration phases are observed at the afternoon, when the hydraulic as well as the nitrogen loads reach the 2nd stage.

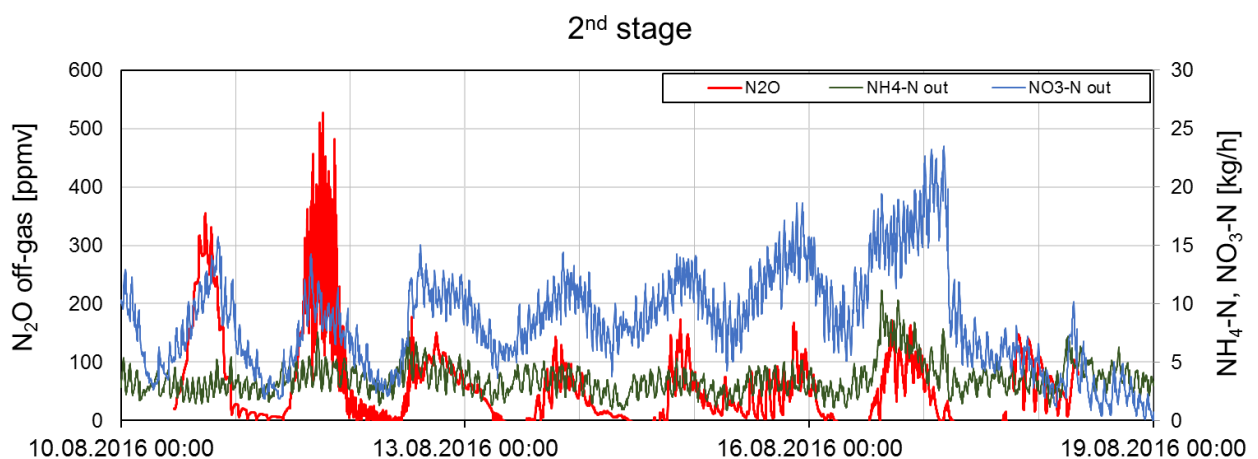


Figure 58: Correlation of the N₂O off-gas concentrations and the ammonia and nitrate output load from the monitored aeration tank 2 in the 2nd stage on WWTP Kirchbichl

Figure 58 depicts the correlation of the N₂O off-gas concentration and the ammonia and nitrate output loads of the 2nd stage and clearly shows, that intensive nitrification (higher NO₃-N load) is accompanied by an increase of the N₂O off-gas concentration.



The higher $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ loads observed on 17./18.08.2016, combined with a lower measured O_2 concentration in the aeration tank (0.5 mg/l) indicate a higher influent load, which was confirmed from the operating log sheets of the WWTP.

The off-gas measurements from the SDE treatment tank was carried out from 18.08.2016 till 31.08.2016. The results are depicted in Figure 59. Since the loading of the tank occurs directly after dewatering, without intermediate storage of the effluent, the online measured $\text{NH}_4\text{-N}$ concentration in the tank correlates with the operating time of the centrifuges. Once the dewatering is started, the continuous measured $\text{NH}_4\text{-N}$ concentration in the tank increases until the dewatering is finished and reaches a value of approx. 500 mg/l due to the dilution in the tank. Subsequently the $\text{NH}_4\text{-N}$ load is nitrified and the $\text{NH}_4\text{-N}$ concentration as well as the pH level decreases. In case a critical pH value is reached ($\text{pH}\approx 6$), the nitrification is limited and the $\text{NH}_4\text{-N}$ concentration remains constant even when the O_2 concentration exceeded 5 mg/l (18.08.2016). The decrease of the N_2O off-gas concentration indicates a higher N_2O stripping compared to N_2O production, which results from the feeding strategy (replacing the treated, N_2O -rich SDE with fresh one). When SDE is pumped in the tank, the O_2 concentration decreases due to the higher oxygen consumption for ammonia oxidation. Once the feeding is turned off, the O_2 concentration raises again to the initial level of 2 mg O_2 /l. The temperature in the SDE treatment tank correlates well with the loading rate and varies between 28.5°C and 31.0°C. For the longer interval without feeding (24.-26.08.2016) N_2O off-gas concentration of approx. 1300 ppm was reached. The stable concentration let assume, that the produced and stripped N_2O were equal. Once the treatment tank was loaded again, the produced N_2O in the liquid phase was displaced with the effluent of the tank to the 1st stage.

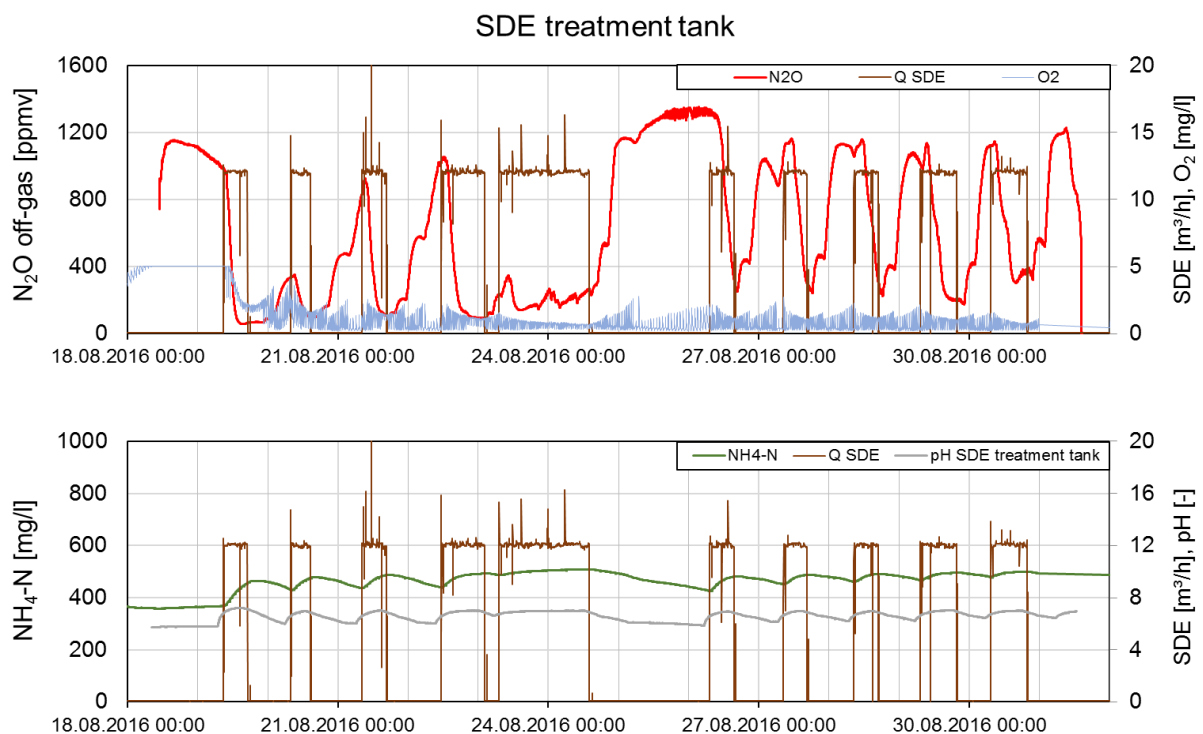


Figure 59: N_2O off-gas concentration, O_2 and ammonia concentration as well as pH in the SDE treatment tank correlated with the SDE

The measurement in the 1st stage shows a stable N₂O off-gas concentration during the anoxic phases. As shown in Figure 60, the tank was aerated for 30 min, at the end of the SDE discharging, when the effluent of the SDE treatment tank was returned to the 1st stage. Immediately after starting the aeration the N₂O off-gas concentration increased due to stripping of dissolved N₂O. The depth of the tank (5 m) is assumed to be sufficient for the fine air bubbles to ensure equilibrium between the dissolved and gaseous compounds.

For the measured maxima of approx. 1300 ppmv in the off-gas at temperature of 30°C, approx. 1500 µg N₂O_{dissolved}/l were calculated, based on the Henry-Dalton's Law. Returning this partial stream to the 1st stage, a dilution due to mixing with the WWTP influent (after the sand trap) occurs, where theoretically N₂O concentrations of 27 µg N₂O_{dissolved}/l are expected. Since the measured N₂O off-gas concentration was much higher compared to the calculated (15 ppmv) for the maximal concentration of the treated SDE, an additional N₂O input load for the 1st stage was assumed (e.g. SK 2 or WWTP's influent). Considering the dilution of SK 2 (theoretical N₂O concentration of 190 µg N₂O/l), the calculated N₂O concentration of the WWTP's influent amounts to 65 µg N₂O/l.

The WWTP influent contains activated sludge, which originates from the biological stage of indirect discharger. Hence, in the sewer system a biofilm is also present, a nitrogen conversion (nitrification and denitrification processes) can also be assumed. The WWTP's data for the influent characteristics show a variation of the NO₃-N daily average concentrations in the range 0.7-6.9 mg/l (average of 4.2 mg/l) as well as lower BOD and COD concentrations as on other WWTPs and confirm at least partially nitrification and/or denitrification in the sewer system. Due to the limited data regarding the influent (no NO₂-N concentration) it is difficult to evaluate the completeness of the processes occurring in the raw sewage. Hence, the estimated N₂O concentration of 65 µg/l is within the range of the literature reported data (up to 45 µg N₂O/l (Short et al., 2014) resp. approx. 100 µg N₂O/l in primary clarifiers (Rodriguez-Caballero et al., 2014)), this value was accepted and the higher N₂O off-gas concentrations were lead back primarily to the influent.

According to Anthonisen et al. (1976) the measured NO₂-N concentration of approx. 500 mg/l corresponds to a FNA concentration of max. 5.4 mg HNO₂/l under the operational conditions (pH_{min}=5.8; T_{min}=28°C). This concentration was lower than the reported in Law et al., 2011 (4.6 mg HNO₂-N/l or 15.4 mg HNO₂/l), therefore no inhibition of the acclimated sludge is expected.

Once the aeration was turned off, the N₂O off-gas concentration increased until reaching the gas-liquid equilibrium and decreased slightly (only 10 %) up to the end of the measurement. In contrast to the observed N₂O variations in the SDE treatment tank and the 2nd stage, no daily variation was observed in the 1st stage, when the tank is not aerated. This fact confirms the assumption, that N₂O emissions from not aerated tanks or zones can be neglected due to the very slow gas exchange between atmosphere and liquid over the water surface.

Based on the N₂O concentrations and the aeration flow rates N₂O emission loads for each treatment tank will be estimated as the next step.



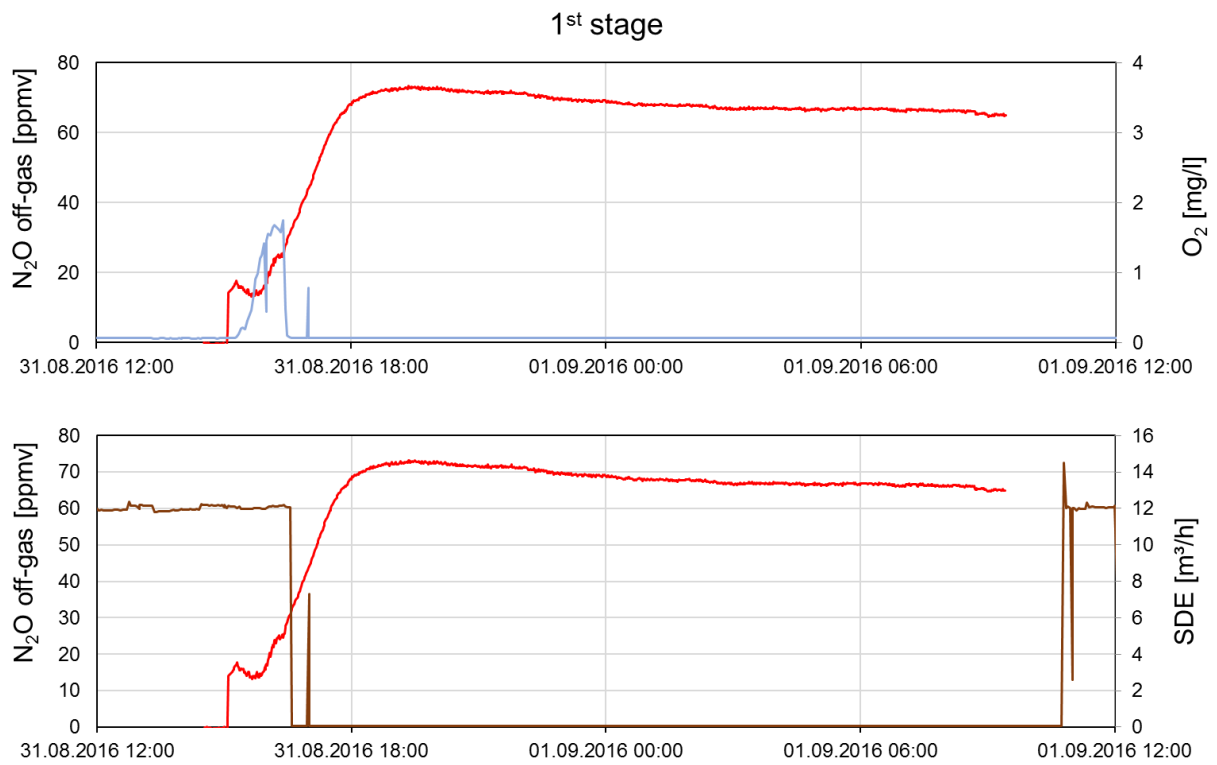


Figure 60: N₂O off-gas concentration, O₂ concentration in the 1st stage as well as flowrate of the directed SDE treatment tank effluent into the 1st stage at WWTP Kirchbichl

6. Outlook

As the WWTP Kirchbichl is situated in a winter tourism region and the wastewater treatment performance of the modified operation (POWERSTEP concept) can be proved under unfavorable conditions in full-scale (low temperature, high ammonia load, low C/N ratio), a second start-up phase of the SDE treatment tank is planned. The sampling frequency will be increased for detailed recording of the operational change. The new start-up phase will be accompanied by further N₂O measurements. The N₂O emissions and the N₂O emission factors of each biological step as well as of the entire WWTPs will be determined.

To investigate optimal operation and possible limitation factors, further laboratory trials will be realized according to the needs. Based on results from the full-scale implementation and the collected energy data, the dynamic model for the two-stage treatment plant will be further developed. The findings of this innovative SDE management shall be extended as well to one-stage treatment plants. Additionally, special emphasis will be placed on the modelling of the energy system which will be the basis for the intended energy optimized WWTP and the comparative energetic assessment of the selected ammonium removal options, as well as N₂O monitoring for the LCA assessment (WP 5). The aim will be a decision support tool using few simple key factors depending on the WWTP's characteristics for finding the appropriate resource and energy optimized SDE treatment technology.



7. References

- Abeling, U. (1994) **Stickstoffelimination aus Industrieabwässern, Denitrifikation über Nitrit.** Veröffentlichung des Institutes für Siedlungswasserwirtschaft und Abfalltechnik der Universität Hannover. Heft 86.
1. AEV für kommunales Abwasser. **Verordnung des Bundesministers für Land- und Forstwirtschaft über die Begrenzung von Abwasseremissionen aus Abwasserreinigungsanlagen für Siedlungsgebiete.** BGBl. Nr. 210/1996
- Ahn, J. H., Kim, S., Park, H., Rahm, B., Pagilla, K. and Chandran, K. (2010). **N₂O Emissions from Activated Sludge Processes, 2008–2009: Results of a National Monitoring Survey in the United States.** Environmental Science & Technology 44(12): 4505-4511.
- Ali, M., Rathnayake, M.L.D., Zhang, L., Ishii, S., Kindaichi, T., Satoh, H., Toyoda, S., Yoshida, N., Okabe, S. (2016). **Source identification of nitrous oxide emission pathways from a single-stage nitrification-anammox granular reactor.** Water research 102: 147-157.
- Anthonisen, A.c., Loehr, R.c., Prakasam, T.B.s., SRINATH, E.G. (1976). **Inhibition of nitrification by ammonia and nitrous acid.** Water Pollution Control Federation 48(5):835-52.
- Baumgartner, T., Svardal K. (2014) **EOS-Energieoptimierung Schlammbehandlung, Trübwasserbehandlung – Versuchsanlage.** Technical Report
- Buday, J., Drtil, M., Hutnan, M. and Derco, J. (1999) **Substrate and Product Inhibition of Nitrification.** Chemical papers – Chemické zvesti 53(6): 379-383
- Castro-Barros., C.M., Rodriguez-Caballero, A., Volcke, E.I.P., Pijuan, M. (2016). **Effect of nitrite on the N₂O and NO production on the nitrification of low-strength ammonium wastewater.** Chemical Engineering Journal 287: 269-276.
- Daelman, M. R. J., De Baets, B., van Loosdrecht, M. C. M. and Volcke, E. I. P. (2013). **Influence of sampling strategies on the estimated nitrous oxide emission from wastewater treatment plants.** Water Research 47(9): 3120-3130.
- Daigger, G, Sanjines, P., Pallansch, K., Sizemore, J., Wett, B. (2011) **Implementation of a Full-Scale Anammox-Based Facility to Treat an Anaerobic Digestion Sidestream at the Alexandria Sanitation Authority Water Resource Facility.** Water Practice & Technology 6 (2)
- Demirer and Othman (2009) **Removal of ammonium and phosphate from the supernatant of anaerobically digested waste activated sludge by chemical precipitation.** Bioresource Technology 100 (13), 3236–3244
- Desloover, J., De Clippeleir, H., Boeckx, P., Du Laing, G., Colsen, J., Verstraete, W. and Vlaeminck, S. E. (2011). **Floc-based sequential partial nitritation and anammox at full scale with contrasting N₂O emissions.** Water Research 45(9): 2811-2821.
- Doyle, J., Parsons, S.A. (2002) **Struvite formation, control and recovery.** Water Research 36, 3925–3940.
- eDAB-**elektronisches Daterfassungs-Analyse und Bilanzierungstool für Abwasserreinigungsanlagen.** www.edab.at/
- Foley, J., de Haas, D., Yuan, Z. and Lant, P. (2010). **Nitrous oxide generation in full-scale biological nutrient removal wastewater treatment plants.** Water Research 44(3): 831-844.
- Frey, W., 1989. **Ermittlung des Sauerstoffzufuhrvermögens mit einer Gasbilanz.** Technische Hochschule Darmstadt, Schriftenreihe WAR, Band 36, 77-84.
- Fux, C., Velten, S., Carozzi, V., Solley, D., Keller, J. (2006) **Efficient and stable nitritation and denitrification of ammonium-rich sludge dewatering liquor using an SBR with continuous loading.** Water Research 40(14), 2765–2775
- Gabarró, J., Gonzalez-Carcamo, P., Ruscalleda, M., Ganique, R., Gich, F., Balaquer, M.D., Colprim, J. (2014). **Anoxic phases are the main N₂O contributor in partial nitritation reactors treating high nitrogen loads with alternate aeration.** Bioresource Technology 163: 92-99.
- Geijsbjerg, B., Frette, L. and Westermann, P. (1998). **Dynamics of N₂O production from activated sludge.** Water Research 32(7): 2113-2121.



- Gustavsson, D. J. I. and la Cour Jansen, J. (2011). **Dynamics of nitrogen oxides emission from a full-scale sludge liquor treatment plant with nitrification**. Water Sci. Technol. 63(12): 2838-2845.
- Gustavsson, D. (2011) **Nitrification and Denitrification in sludge liquor treatment**. Thesis. Water and Environmental Engineering Department of Chemical Engineering Lund University, Sweden.
- Guštin, S., Marinšek-Logar, R. (2011) **Effect of pH, temperature and air flow rate on the continuous ammonia stripping of the anaerobic digestion effluent**. Process Safety and Environmental Protection, Volume 89(1), 61–66.
- Hanaki, K., Hong, Z. and Matsuo, T. (1992). **Production of Nitrous Oxide Gas during Denitrification of Wastewater**. Water Sci. Technol. 26(5-6): 1027-1036.
- Hippen, A. (2001) **Einsatz der Deammonifikation zur Behandlung hoch stickstoffhaltiger Abwässer**. Veröffentlichung des Institutes für Siedlungswasserwirtschaft und Abfalltechnik der Universität Hannover. Heft 116
- Joss, A., Salzgeber, D., Eugster, J., König, R., Rottermann, K., Burger, S., Fabijan, P., Leumann, S., Mohn, J., Siegrist, H. (2009). **Full-Scale Nitrogen Removal from Digester Liquid with Partial Nitrification and Anammox in One SBR**. Environ. Sci. Technol. 43: 5301-5306
- Kampschreur, M. J., van der Star, W. R. L., Wienders, H. A., Mulder, J. W., Jetten, M. S. M. and van Loosdrecht, M. C. M. (2008). **Dynamics of nitric oxide and nitrous oxide emission during full-scale reject water treatment**. Water Research 42(3): 812-826.
- Kampschreur, M., Kleerebezem, R., de Vet, W.W.J.M., van Loosdrecht, M.C.M. (2011). **Reduced iron induced nitric oxide and nitrous oxide emissions**. Water Research 45: 5945-5952.
- Kimochi, Y., Inamori, Y., Mizuochi, M., Xu, K.-Q. and Matsumura, M. (1998). **Nitrogen removal and N₂O emission in a full-scale domestic wastewater treatment plant with intermittent aeration**. Journal of Fermentation and Bioengineering 86(2): 202-206.
- Jardin N., Hennerkes J. (2012) **Full-scale experience with the deammonification process to treat high strength sludge water – a case study**. Water Sci. Technol. 65 (3) 447-455
- Lackner, S., Gilbert, E.M., Vlaeminck, S. E., Joss, A., Horn, H., von Loosdrecht, M.C.M. **Full-scale partial nitrification/anammox experiences - An application survey**. Water Research 55, 292-303.
- Lackner S., Thoma K., Gilbert E., Gander W., Schreff D. and Horn H. (2015) **Start-up of a full-scale deammonification SBR-treating effluent from digested sludge dewatering**. Water Sci. Technol. 71 (4) 553-559
- Law, Y., Lant, P. and Yuan, Z. (2011). **The effect of pH on N₂O production under aerobic conditions in a partial nitrification system**. Water Research 45(18): 5934-5944.
- Law, Y., Ni, B.-J., Lant, P. and Yuan, Z. (2012). **N₂O production rate of an enriched ammonia-oxidising bacteria culture exponentially correlates to its ammonia oxidation rate**. Water Research 46(10): 3409-3419.
- Lei, X., Sugiura, N., Feng, C., Maekawa, T. (2007) **Pretreatment of anaerobic digestion effluent with ammonia stripping and biogas purification**. Journal of Hazardous Materials 145 (3), 391–397.
- Lim, Y. and Kim, D.-J. (2014). **Quantification method of N₂O emission from full-scale biological nutrient removal wastewater treatment plant by laboratory batch reactor analysis**. Bioresource Technology 165: 111-115.
- Lotito, A. M., Wunderlin, P., Joss, A., Kipf, M. and Siegrist, H. (2012). **Nitrous oxide emissions from the oxidation tank of a pilot activated sludge plant**. Water Research 46(11): 3563-3573.
- Magrí, A., Béline F., Dabert, P. (2013) **Feasibility and interest of the anammox process as treatment alternative for anaerobic digester supernatants in manure processing – An overview**. Journal of Environmental Management 131, 170–184.
- Mampaey, K., De Kreuk, M., van Dongen, U.G.J.M., van Loosdrecht, M.C.M., Volcke, E.I.P. (2016). **Identifying N₂O formation and emissions from a full-scale partial nitrification reactor**. Water Research 88: 575-585.



- Mouilleron, I., Hyde, K., Reid, K., Keegan, A., Rinck-Pfeiffer, R., Krampe, J., van den Akker, B. (2014) **Start-up of a demonstration-scale deammonification reactor at Bolivar WWTP**. Results of a demonstration trial at a wastewater treatment plant in South Australia. Technical Papers. Water.
- Ni, B.-J., Peng, L., Law, Y., Guo, J., Yuan, Zh. (2014). **Modeling of Nitrous Oxide Production by Autotrophic Ammonia-Oxidizing Bacteria with Multiple Production Pathways**. Environmental Science & Technology 48: 3916-3924.
- Nowak, O., Franz, A., Svardal, K., Müller, V., Kühn, V. (1999). **Parameter estimation for activated sludge models with the help of mass balances**. Wat. Sci. Techn., Vol 39 No.4, pp.113-120.
- Otte, S., Grobbs, N. G., Robertson, L. A., Jetten, M. S. M, Kuenen, J. G. (1996). **Nitrous Oxide Production by *Alcaligenes faecalis* under Transient and Dynamic Aerobic and Anaerobic Conditions**. Applied and Environmental Microbiology 62(7): 2421-2426.
- Pan, Y., Ni, B.-J., Bond, P. L., Ye, L. and Yuan, Z. (2013). **Electron competition among nitrogen oxides reduction during methanol-utilizing denitrification in wastewater treatment**. Water Research 47(10): 3273-3281.
- Pan, Y., Ye, L., Ni, B.-J. and Yuan, Z. (2012). **Effect of pH on N_2O reduction and accumulation during denitrification by methanol utilizing denitrifiers**. Water Research 46(15): 4832-4840.
- Park, K. Y., Inamori, Y., Mizuochi, M. and Ahn, K. H. (2000). **Emission and control of nitrous oxide from a biological wastewater treatment system with intermittent aeration**. Journal of Bioscience and Bioengineering 90(3): 247-252.
- Parravicini, V., Valkova, T., Haslinger, J., Saracevic, E., Winkelbauer, A., Tauber, J., Svardal, K., Hohenblum, Ph., Clara, M., Windhofer, G., Pazdernik, K., Lampert, Chr. (2015). **Reduktionspotential bei den Lachgasemissionen aus Kläranlagen durch Optimierung des Betriebes**. Bundesministerium für Land und Forstwirtschaft, Umwelt und Wasserwirtschaft.
- Peng, L., Ni, B.-J., Eler, D., Ye, L. and Yuan, Z. (2014). **The effect of dissolved oxygen on N_2O production by ammonia-oxidizing bacteria in an enriched nitrifying sludge**. Water Research 66: 12-21.
- Peng, L., Ni, B.-J., Ye, L. and Yuan, Z. (2015). **The combined effect of dissolved oxygen and nitrite on N_2O production by ammonia oxidizing bacteria in an enriched nitrifying sludge**. Water Research 73: 29-36.
- Peng, L., Ni, B.-J., Ye, L. and Yuan, Z. (2015). **N_2O production by ammonia oxidizing bacteria in an enriched nitrifying sludge linearly depends on inorganic carbon concentration**. Water Research 74: 58-66.
- Pijuan, M., Tora, J., Rodriguez-Caballero, A., Cesar, E., Carrera, J., Perez, J. (2014). **Effect of process parameters on nitrous oxide emissions from a nitrification reactor treating reject water**. Water Research 49: 23-33.
- Rodriguez-Caballero, A., Ribera, A., Balcázar, J. L. and Pijuan, M. (2013). **Nitrification versus full nitrification of ammonium-rich wastewater: Comparison in terms of nitrous and nitric oxides emissions**. Bioresource Technology 139: 195-202.
- Rodriguez-Caballero, A., Aymerich, I., Poch, M., Pijuan M., (2014). **Evaluation of process conditions triggering emissions of green-house gases from a biological wastewater treatment system**. Science of the Total Environment 493: 384-391.
- Schneider, Y., Beier, M. and Rosenwinkel, K.-H. (2014). **Influence of operating conditions on nitrous oxide formation during nitrification and nitrification**. Environmental Science and Pollution Research 21(20): 12099-12108.
- Schönharting, B., Rehner, R., Metzger, J.W., Krauth, K., Rizzi, M. (1998). **Release of nitrous oxide (N_2O) from denitrifying activated sludge caused by H_2S -containing wastewater: Quantification and application of a new mathematical model**. Wat. Sci. Tech. 36 (1): 237-246.
- Short, M. D., Daikeler, A., Peters, G. M., Mann, K., Ashbolt, N. J., Stuetz, R. M. and Peirson, W. L. (2014). **Municipal gravity sewers: An unrecognised source of nitrous oxide**. Science of The Total Environment 468-469: 211-218.



- Soler-Jofra, A., Stevens, B., Hoekstra, M., Picioreanu, C., Sorokin, D., Loosdrecht, M.C.M., Perez, J. (2016). **Importance of abiotic hydroxylamine conversion on nitrous oxide emissions during nitrification of reject water.** Chemical Engineering Journal 287: 720-726.
- Strous, M., Heijnen, J. J., Kuenen, J. G. u. Jetten, M. S. M. (1998). **The sequencing batch reactor as a powerful tool for the study of slowly growing anaerobic ammonium-oxidizing microorganisms.** Appl. Microbiol. Biotechnology 50, 589-596
- Svardal, K., Nowak, O., Schweighofer, P. (1998). **Datendokumentation und Auswertung-Plausibilitätsanalyse von Meßwerten.** Wiener Mittellungen, Band 147, 439-475.
- Wicht, H. and Beier, M. (1995). **N₂O-Emissionen aus nitrifizierenden und denitrifizierenden Kläranlagen.** Korrespondenz Abwasser (3): 404-413.
- Thörn, M. and Sörensson, F. (1996). **Variation of nitrous oxide formation in the denitrification basin in a wastewater treatment plant with nitrogen removal.** Water Research 30(6): 1543-1547.
- Twachtmann, U. (1999) **Autotrophe Stickstoff-elimination in Tropfkörpern.** Dissertation. Forschungs- und Entwicklungsinstitut für Industrie- und Siedlungswasserwirtschaft sowie Abfallwirtschaft e. V., Stuttgart. Stuttgarter Bericht zur Siedlungswasserwirtschaft. Band 158. Oldenbourg Industrieverlag GmbH.
- Ye, L., Ni, B.-J., Law, Y., Byers, C. and Yuan, Z. (2014). **A novel methodology to quantify nitrous oxide emissions from full-scale wastewater treatment systems with surface aerators.** Water Research 48: 257-268.
- Wandl, G., Müller-Rechberger, H., Matsché, N., Svardal, K., Winkler, S. (2002). **Two stage activated sludge plants-influence of different operational modes on sludge bulking and nitrification.** Water Sci. Technol. 46 (1-2): 479-486.
- Wang, Q., Jiang, G., Ye, L., Pijuan, M. and Yuan, Z. (2014). **Heterotrophic denitrification plays an important role in N₂O production from nitrification reactors treating anaerobic sludge digestion liquor.** Water Research 62: 202-210.
- Wang, X., Yang, X., Zhang, Z., Ye, X., Kao, C. M. and Chen, S. (2014). **Long-term effect of temperature on N₂O emission from the denitrifying activated sludge.** Journal of Bioscience and Bioengineering 117(3): 298-304.
- Wunderlin, P., Mohn, J., Joss, A., Emmenegger, L. and Siegrist, H. (2012). **Mechanisms of N₂O production in biological wastewater treatment under nitrifying and denitrifying conditions.** Water Research 46(4): 1027-1037.
- Wunderlin, P., Lehmann, M. F., Siegrist, H., Tuzson, B., Joss, A., Emmenegger, L. and Mohn, J. (2013). **Isotope Signatures of N₂O in a Mixed Microbial Population System: Constraints on N₂O Producing Pathways in Wastewater Treatment.** Environmental Science & Technology 47(3): 1339-1348.
- Zheng, H., Hanaki, K. and Matsuo, T. (1994). **Production of nitrous oxide gas during nitrification of wastewater.** Water Sci. Technol. 30(6): 133-141.

