Strategies for Achieving Energy Neutrality in Biological Nutrient Removal Systems – a Case Study of the Slupsk WWTP (northern Poland)

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ABSTRACT

The paper presents a model-based evaluation of technological upgrades on the energy and cost balance in a large biological nutrient removal wastewater treatment plant (WWTP) in the city of Slupsk (northern Poland). The proposed upgrades include chemically enhanced primary sludge removal and reduction of the nitrogen load in the deammonification process employed for reject water treatment. Simulations enabled to estimate the increased biogas generation and decreased energy consumption for aeration. The proposed upgrades may lead the studied WWTP from the energy deficit to energy neutrality and positive cost balance, while still maintaining the required effluent standards for nitrogen. The operating cost balance depends on the type of applied coagulants/flocculants and specific costs of electric energy. The choice of the coagulant/flocculent was found as the main factor determining a positive cost balance.

KEYWORDS

BNR plant, biogas production, cost balance, energy balance, energy neutrality, modelling.

INTRODUCTION

Biological nutrient removal (BNR) wastewater treatment plants (WWTPs) are energy intensive facilities. The specific electric energy consumption in different European countries generally ranges from 0.36 to 0.67 kWh per cubic metre of treated wastewater (Hernández-Sancho *et al*. 2011). The amount of energy needed for operations varies widely among individual treatment plants depending on the actual wastewater flow rate and characteristics, treatment technology, required effluent quality and sludge disposal. It is expected that the energy consumption will grow over time due to a number of factors, such as growth of wastewater volume and the contaminant load to be treated, as well as increasingly stringent regulatory and environmental protection standards. In order to meet the current and future requirements, many existing Polish WWTPs need to be extended or/and upgraded in order to comply with those requirements. Despite the progress in wastewater management, there are still challenges in the range of environmental protection. Optimization of energy consumption, efficiency of design and of equipment and technology operations, energy recovery processes, and good management of energy pricing are being increasingly considered in the field of wastewater treatment. A higher energy efficiency means lower energy consumption, lower greenhouse gases emissions, and lower operating costs for WWTPs (Hernández-Sancho *et al*. 2011).

Regardless of a WWTP size, most of the energy is consumed during biological treatment. Aeration of bioreactors may account for 50%-75% of the total electric energy consumption in WWTPs (Gori et al. 2011, Jiang and Stenstrom 2012, Wang *et al*. 2013, Zhou *et al*. 2013, Gao *et al*. 2014,). The type of aeration is one of the factors that determine energy efficiency of the plant. The diffused aeration systems are more efficient but also more energy-consuming in comparison with the mechanical aeration systems (Hernández-Sancho *et al*. 2011). The aeration system control is an important tool to minimize energy consumption in activated sludge systems and is closely linked to how the effluent criteria are defined (Åmand and Carlsson 2012). More specific studies showed that the production of ammonium by the digestion process and the lack of available carbon source for denitrification in bioreactors are the limiting factors to the overall energy efficiency (Descoins *et al*. 2012). The characteristics of the carbon substrate were recognized as having a close relationship to energy conservation in the conventional activated sludge systems (Wang *et al*. 2013).

Recently, a new paradigm in wastewater treatment is widely disseminated around the world. Although, the efficient treatment is undoubtedly the primary objective of the WWTPs, the plants should also be designed with respect to resources and energy recovery. The energy balance in WWTPs can be improved by either decreasing energy consumption or increasing the share of renewable energy in covering the energy demand. The new approach emphasizes a movement towards energy neutral or even energy positive wastewater treatment, as well as importance of separating nitrogen and organic waste streams to maximize energy capture (Gao *et al*. 2014). The separation of organic matter from wastewater influent can enhance biogas production, while the separated nitrogen stream can be treated with the energy efficient deammonification processes with no organic carbon demand. Jenicek *et al*. (2012) found that the self-sufficiency depends on the optimization of the total energy consumption of the plant, and an increase in the specific biogas production from sewage sludge.

Chemically enhanced primary wastewater treatment (CEPT) and the deammonification process give an opportunity for reduction of energy consumption (Kroiss and Cao 2014). Partial nitrification/anammox (deammonification) process is the promising technology of nitrogen removal. An important benefit of partial nitrification is the reduced oxygen demand for nitrification by up to 25% (Bournazou *et al*. 2013). Application of the anammox process in the main stream may reduce energy consumption by 45% compared to the conventional nitrification-denitrification (Kartal *et al*. 2010). Ammonia-rich anaerobic digester liquors (15-20% of the inlet ammonia load) can be treated with the very economic autotrophic nitritation/anammox process requiring half of the aeration energy and no organic carbon source compared to the conventional nitrification-denitrification (Siegrist *et al*. 2008). The introduction of that technology allows to increase a hydraulic retention time (HRT) in the primary clarifier to improve biogas production and reduce aeration energy for COD removal and nitrification at similar overall N-removal (Siegrist *et al*. 2008, Carrère *et al*. 2010).

Primary sludge tanks removal efficiencies vary from 40% to 60% for TSS and from 25% to 40% for COD. By adding chemicals these efficiencies can be enhanced to about 80 to 90% for TSS and from 50% to 70% for COD removal (Kroiss and Cao 2014). The concept based on a maximum extraction of organic matter into the sludge via coagulation, flocculation and microsieving leads to the increased energy recovery in anaerobic sludge digestion and decreased aeration demand for carbon mineralisation (Remy *et al*. 2014). The coagulation and flocculation are the proved techniques of primary sludge separation, while the microsieving is an option to separate the sludge with high methane potential instead of primary clarifiers.

The former studies on model-based examination of WWTPs demonstrated how models can significantly improve the evaluation, design and operation of those facilities. The case studies presented the use of modelling for cost-effective planning, plant operation optimization, as well as model limitations (Phillips *et al*. 2009). The preferred plant-wide and resource recovery modelling requires a primary settler model. The typical organic fractions are modified by primary treatment, which subsequently affects the downstream processes (Bachis *et al*. 2015). Particulate COD removal during primary sedimentation implies a lower energy demand on the subsequent secondary treatment, as well as increase in the biogas production and associated energy recovery (Gori *et al*. 2013). Another study showed a potential for reduction of the energy consumption through operational changes only, without compromising the current effluent quality (Puchongkawarin *et al*. 2015).

The aim of the paper is to present a model-based evaluation of technological upgrades on the energy balance in a large BNR WWTP in the city of Slupsk (northern Poland). The proposed upgrades include chemically enhanced primary sludge removal and reject water treatment with the deammonification process. Simulations enabled to estimate the increased biogas generation and decreased energy consumption for aeration. It was shown that the proposed upgrades may lead the studied WWTP from the energy deficit to energy neutrality.

MATERIAL AND METHODS

Plant characteristics and collection of the operational data

The studied BNR WWTP is a large facility treating on average 20,000-25,000 m^3 d^{-1} (5.3-6.6 MGD) of wastewater from the city of Slupsk and surrounding communities. The designed capacity of the plant is 200,000 PE (population equivalent). A simplified process schematic diagram presenting the idea of the plant is shown in Figure 1. The actual scheme was extended with the potential upgrades described further in the paper.
Suspended solids are removed from raw wastewater in radial, horizontal primary clarifiers located at the head of the main stream. There are three parallel wastewater trains for BNR processes. The plant applies the Bardenpho process configuration including anaerobic and anoxic compartments, followed by an aerobic compartment and secondary clarifiers. The system is flexible and allows to alter the anaerobic/anoxic and anoxic/aerobic conditions in the selected compartments. The mixed liquor recirculation (MLR) may be directed from the aerobic (nitrification) compartment to one or more of the preceding compartments. Air is supplied by a fine bubble membrane diffusers system supplied by four blowers, each of 160 kW of the rated power. The external circulation returns waste activated sludge (WAS) from the secondary clarifier to the tank mixing the flow with raw wastewater after sedimentation step. Wastewater effluent is discharged to the local river (Slupia) running to the Baltic Sea. At present no chemical-aided processes are carried out at the plant.

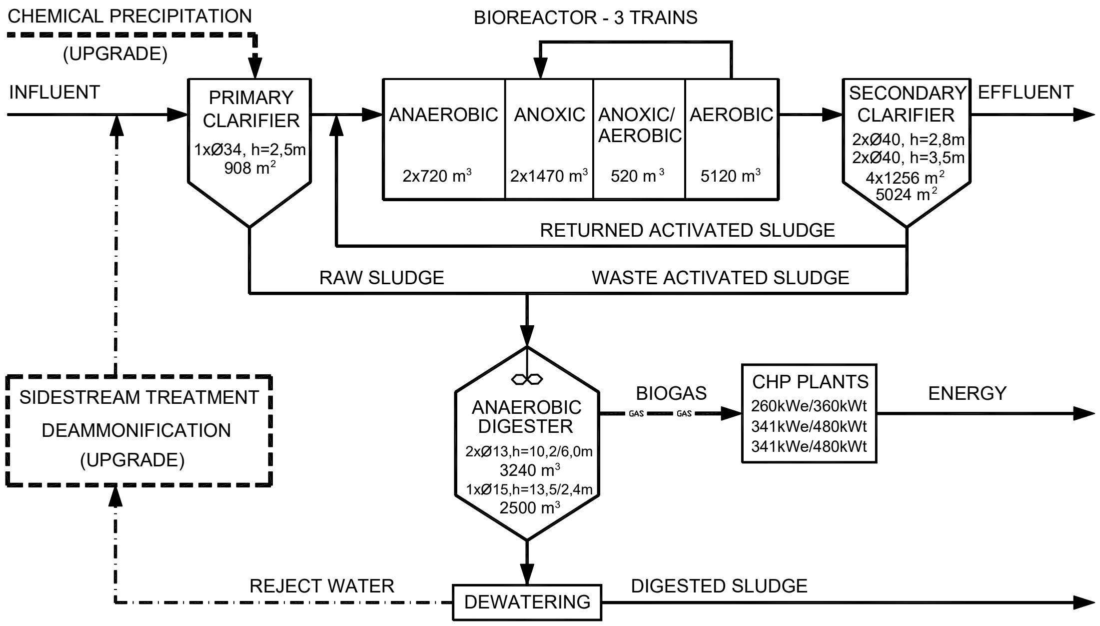

Figure 1. Schematic diagram presenting the idea of the plant including the proposed upgrades (dashed line)

Historical data sets were collected as routine measurements taken in the operational terms in 2013. A set of basic parameters related to raw wastewater influent and effluent is reported in Table 1. The annual average values and for summer period are listed separately since they refer to the model calibration in modelling stage of the study. The average values for summer period are close to the annual average values except for temperature, total nitrogen (TN) and total phosphorus (TP) in wastewater effluent. The annual results were affected by winter period with worse results for TN and better results for TP. The contaminants load expressed in population equivalent (1 PE = 60 g BOD_5 d^{-1} per capita) was approximately as high as 249,500 PE in the studied year.

Table 1. Wastewater characteristics of the studied plant

Parameter	**Flow rate**	**Temperature**	**TSS**	**COD**	**TN**	**TP**
	m^3 d^{-1}	**°C**	**g m^{-3}**	**g COD m^{-3}**	**g N m^{-3}**	**g P m^{-3}**
			Wastewater influent			
Summer average ±SD	21889 ±4023	19.1 ±1.0	434 ±86	1039 ±189	82 ±9	12 ±2
Number of samples	92	92	39	39	25	39
Annual average ±SD	23572 ±4128	14.6 ±3.7	449 ±134	1131 ±256	82 ±13	12 ±3
Number of samples	365	365	146	146	94	146
			Wastewater effluent			
Summer average ±SD	-	-	9.5 ±6.6	33.1 ±4.8	9.7 ±1.6	0.7 ±0.3
Number of samples	-	-	39	39	27	39
Annual average ±SD			9.3 ±6.3	33.7 ±5.0	11.8 ±3.7	0.4 ±0.3
Number of samples	-	-	237	237	237	237

Sludge handling processes include thickening and anaerobic digestion. The primary sludge thickener is a gravity type while centrifuges and drum thickeners are used for waste activated sludge (WAS). Approximately 30% of WAS supernatant after centrifuges is then disintegrated by an ultrasound technology. Table 2 reports the operational characteristics of primary sludge and WAS after the thickening processes. The flow rates, total suspended solids (TSS) and volatile suspended solids (VSS) concentrations represent the values related to the anaerobic digestion feedstock.

Table 2. Primary sludge and WAS characteristics in the studied plant

Parameter	Primary sludge			WAS		
	Flow rate	TSS	VSS	Flow rate	TSS	VSS
	$m^3\ d^{-1}$	%	%	$m^3\ d^{-1}$	%	%
Summer average ±SD	142 ±22	3.2 ±0.8	76.4 ±3.5	95 ±20	5.8 ±1.2	71.3 ±1.7
Number of samples	92	17	17	92	32	16
Annual average ±SD	132 ±28	3.7 ±0.9	80.6 ±5.1	115 ±32	5.5 ±1.0	75.2 ±4.4
Number of samples	365	47	47	365	88	42

The anaerobic digesters with the total volume of 5740 m^3 are fed with the sludge and a small amount of external fat. The process is carried out under mesophilic conditions at temperature in the range of 35-38°C. Each digester is equipped with a propeller and external pump for sludge recirculation through heat exchangers. Digested sludge is dewatered by centrifuges and then composted for agricultural use or dumping. Table 3 reports the operational characteristics of digested sludge. Reject water is returned to the head of the plant, before primary clarifier without any further treatment.

Table 3. Digested sludge characteristics in the studied plant

Parameter	TSS	VSS	pH	VFA	Alkalinity
	%	%	-	$g\ m^{-3}$	$g\ CaCO_3\ m^{-3}$
Summer average ±SD	3.2 ±0.3	67.8 ±2.5	7.2 ±0.1	56 ±40	3416 ±438
Number of samples	75	42	90	78	90
Annual average ±SD	3.1 ±0.4	69.7 ±3.7	7.2 ±0.1	63 ±51	3763 ±542
Number of samples	141	141	292	271	292

Biogas generated in the anaerobic digesters (4430 $m^3\ d^{-1}$) contains 60-65% of methane, carbon dioxide and trace amounts of hydrogen sulfide. Biogas conditioning processes include H_2S removal in a bed filled with active ferric compounds, drying as well as silica removal. The conditioned gas is then used for energy generation in combined heat and power (CHP) plants. The plant characteristics are given in Fig.1. Alternatively biogas can be burned by torch or in a gas boiler. On the other hand, the CHP plant engines can also be driven with natural gas. The annual production of electric energy in biogas cogeneration was nearly 2988 MWh in the studied year, equivalent to 8185 kWh d^{-1} on average. In the summer period, the average daily energy production (6659 kWh d^{-1}) was lower in comparison with the annual average value. Electric energy from cogeneration or power grid was distributed to the receivers for technological and other purposes. The total energy consumption at the studied WWTP was 4091 MWh (11207 kWh d^{-1}). A part of the annual energy produced in cogeneration (105 MWh) was sold and supplied to the external grid.

General organization of experimental and modelling procedure

The purpose of experimental part of the study was to prepare data for the modelling procedure to enable the model calibration and validation, and ultimately run simulations for the strategies being the subject of the study. The links between the experimental data and the modelling procedure are presented in Table 4.

Table 4. Links between the experimental data and the modelling procedure

Source of data	Step in the modelling procedure
Laboratory-scale two-phase batch experiments	Biological reactor model calibration
Full-scale measurement campaign	Biological reactor model calibration (summer conditions) and validation (winter conditions) under dynamic conditions
Historical operational data and laboratory experiments for suspended solids chemical precipitation	Simulation of the strategies for achieving energy neutrality at steady-state conditions at the plant-wide model

Experimental procedure for the main stream line

The experimental part of the study comprised several laboratory batch tests and a 4-day measurement campaign in the full-scale bioreactor.

The laboratory-scale experiments were carried out in an experimental set-up consisting of two parallel batch reactors (max. volume of 4.0 dm^3), described in details by Czerwionka *et al.* (2012). The reactors were equipped with mechanical stirrers, temperature and dissolved oxygen (DO) control system and a computer. Two kinds of 2-phase experiments were conducted under anaerobic-aerobic and anaerobic-anoxic conditions, with respect to the following processes: PO_4 release/uptake rates, nitrification rates and denitrification rates. The duration of the anaerobic phase was 2.5 h, followed by 4.5 h of aerobic (at the DO set point = 6 g O_2 m^{-3}) or anoxic conditions (after addition of KNO_3). At the beginning of experiment, the process biomass (fresh returned activated sludge (RAS)) was diluted with the settled wastewater. The dilution rate was adjusted to obtain the mixed liquor suspended solids (MLSS) concentration at approximately 2.5 kg m^{-3} in the reactors, but the actual MLSS concentrations were measured at the beginning and at the end of the experiment. Samples of 100-150 cm^3 were collected with the variable frequency, filtered under vacuum pressure on 1.2 μm pore size filter and then analyzed.

The additional 96-hour "continuous" measurement campaign was conducted in the full-scale biological reactor in the studied WWTP. Grab samples were withdrawn every two hours at the following locations: reactor inlet, anaerobic zone, anoxic zone and reactor effluent. The samples were analyzed for several parameters including COD, soluble COD (sCOD), TN, NH_4-N, NO_3-N and PO_4-P.

Experimental procedure for suspended solids chemical precipitation

Composite 24-hour samples of raw wastewater were used to the experiments. The samples were collected before the bar screens of the studied WWTP. Technical pure ferric (III) sulphate $Fe_2(SO_4)_3$ and anionic flocculent A110 from Kemipol Company (Poland) were used in the experiments. The choice of the reagents for the coagulation/flocculation processes was based on the suggestion of the supplier and the experience of WWTPs in Poland. Multiple doses of the reagents were used, including 50, 75 and 100 g m^{-3} of ferric (III) sulphate and 0.2, 1.0, 2.0 and 3.0 g m^{-3} of anionic flocculants. The laboratory experiments with raw wastewater without pretreatment (only after 2-hour sedimentation) and after coagulation/flocculation processes were carried out in order to determine the effects of enhanced solids precipitation. Raw wastewater (1 L) was processed with 2 hours of sedimentation and then the supernatant liquid was decanted. In the stage with coagulants/flocculants use, raw wastewater (1 L) was placed to a glass beakers (1 L) and the specified doses of ferric (III) sulphate were added. The content was rapidly mixed by magnetic stirrer for 30 seconds (400 rpm). Subsequently, specified doses of flocculent were added and the content was mixed slowly for 10 minutes (130 rpm). After stirring, the wastewater was processed with 2-hour sedimentation and then the supernatant liquid was decanted.

Analytical methods

The samples of raw wastewater, supernatant after sedimentation without and with addition of reagents were analyzed on total and soluble COD (TCOD, SCOD), total and volatile suspended solids (TSS, VSS), total phosphorus (TP) and total nitrogen (TN). The samples of mixed liquor (wastewater with activated sludge) were filtered under vacuum pressure through a 1.2 μm pore size Millipore (Billerica MA, USA) nitrocellulose filter before the analysis. TN concentrations were determined using a TOC analyser (TOC-V_{CSH}) coupled with a TN module (TNM-1) (SHIMADZU Corporation, Kyoto, Japan). The concentrations of sCOD, COD, inorganic N forms (NH_4-N, NO_3-N and NO_2-N) and TP were determined using Xion 500 spectrophotometer (Dr Lange GmbH, Berlin, Germany). The analytical procedures, which were adopted by Dr Lange and SHIMADZU Corporation, followed the Standard Methods for Examination of Water and Wastewater (APHA 2005). TSS and VSS were measured by the gravimetric methods in accordance with the standard methods (APHA 2005).

Modelling procedure

A simulator environment GPS-X^{TM} ver. 6.4. (Hydromantis, Inc, Canada) was used in the modelling part of the study. The plant-wide model consisted of separate models dedicated to the following unit processes: modified ASM2d model for activated sludge and MantisAD model for anaerobic digestion. The modifications applied to ASM2d model and used in the present study were described in detail by Swinarski *et al*. (2012). MantisAD is a model developed by the Hydromantis team in order to simplify modelling of anaerobic digestion and dedicated for the facilities with limited amount of data (Copp *et al*. 2005). The process of methane production in MantisAD is depicted by acidogenesis, acetogenesis and methanogenesis utilizing two different degradation pathways. The model assumes that composite organic material is disintegrated to both slowly biodegradable and readily biodegradable particulate material. The particulate material undergoes hydrolysis resulting in the production of slowly and readily biodegradable soluble substrate. In the next step, the soluble material is fermented to acetate and hydrogen and then methane gas is generated. The MantisAD model allows for calibration based on an influent fractionation model and kinetic destruction of particulate material. A simultaneous or iterative approach is used to calculate pH via ion balance.

The plant-wide model layout prepared for the studied plant in the GPS-X software is presented in Figure 2. The icons represent the separate processes models, including suspended solids settling, anaerobic, anoxic and aerobic stages of biological carbon and nutrients removal in the main wastewater line, as well as sludge handling processes with thickening and anaerobic digestion processes. The "external carbon" icon represents an option of external carbon addition to the biological reactor anaerobic/anoxic compartments in the case of a carbon deficit for denitrification. However, this option was not used in the present study. The sidestream treatment line was extended with an object simulating nitrogen removal in the deammonification process. A simplified model was used to asses nitrogen removal efficiency from reject water – the experimental model using proportional correlation between the object influent and effluent. The maximum removal efficiency used in the "Black Box" model represents the anammox process limit of technology (LOT). The enhanced primary sludge removal was simulated by altering the efficiency of suspended solids removal in the primary clarifier (a non-reactive model).

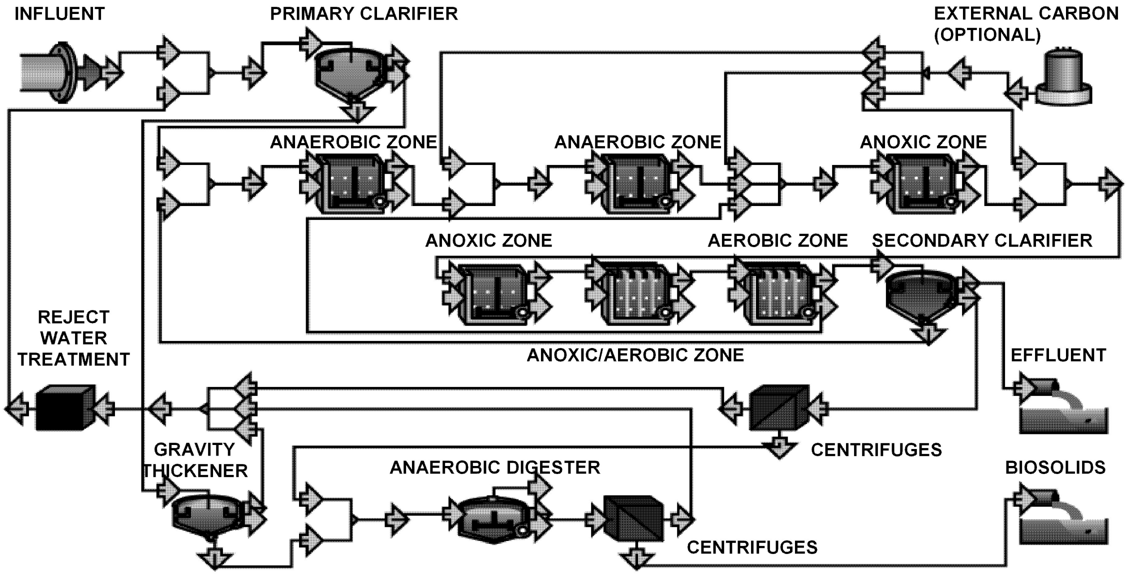

Figure 2. Model layout for the studied WWTP in Slupsk (GPS-X)

The modelling study consisted of three main steps: activated sludge and anaerobic digestion calibration and validation, simulation of the plant-wide model at steady-state conditions, evaluation of strategies for the upgraded technologies, including enhancing primary sludge withdrawal and increasing the efficiency of nitrogen removal in the sidestream treatment line. The ASM2d model calibration was carried out on a partial model including the bioreactor compartments and secondary clarifier. The compartment dimensions (volumes) are given in Figure 1. In the present study, the MLR was directed to the first anoxic zone while the external circulation of RAS from the bottom of the secondary clarifier to the anaerobic zone. Kinetic coefficients adjusted in the present study are listed in the Table5. The coefficients were estimated based on laboratory-scale experiments carried out under summer and winter conditions as well as the full-scale 4-day measuring campaign. The set of the ASM2d kinetic parameters presented in Table 5 refers to the standard temperature of 20°C. The simulations were run at the process temperature of 19°C which is the average temperature for the summer period at the studied WWTP.

The validated biological model was then extended with sludge handling processes and reject water line. The fractions of influent compounds used in steady-state simulations of the extended plant-wide model were adjusted based on the measurements from the experimental stage and measurements carried out in full-scale (in the operating terms). The Influent Advisor GPS-X module was used for that purpose. A set of the assumed fractions and stoichiometric ratios is reported in Table 6.

In the anaerobic digestion modelling procedure, MantisAD default values for kinetic parameters were used, except for hydrolysis rate of readily biodegradable particulate material ($K_{hyd,rb}$ = 0.7 d^{-1} versus default 10 d^{-1}) and half-saturation of acetate uptake (K_{sac} = 170 g COD m^{-3} versus default 150 g COD m^{-3}). The assumed value of $K_{hyd,rb}$ is in the range of 0.3-0.7 g COD m^{-3} as given by Henze *et al*. (2002) for suspended solids hydrolysis in anaerobic processes. The pH and hydrogen solver was used with the set value of pH = 7,2 as the value achieved under the normal operational conditions (Table 3).

Table 5. Set of kinetic coefficients estimated during the modified ASM2d model calibration

Parameter	Symbol	Unit	Default value in ASM2d	Value after calibration
Heterotrophic Biomass (X_H):				
Heterotrophic maximum specific growth rate	μ_H	d^{-1}	6.0	3.0
Oxygen half saturation coefficient	$K_{O,H}$	g O2 m^{-3}	0.2	0.08
Ammonium (as a nutrient) half saturation coefficient	$K_{NH4,H}$	g N m^{-3}	0.05	0.02
Phosphate (as a nutrient) half saturation coefficient	$K_{PO4,H}$	g P m^{-3}	0.01	0.001
Autotrophic Bacteria (X_A):				
Autotrophic maximum specific growth rate	μ_A	d^{-1}	1.0	1.05
Ammonium (as a substrate) half saturation coefficient	$K_{NH4,A}$	g N m^{-3}	1.0	1.2
Phosphate (as a nutrient) half saturation coefficient	$K_{PO4,A}$	g P m^{-3}	0.01	0.001
Poly-P Accumulating Biomass PAO (X_{PAO}):				
Rate constant for storage of PHA	q_{PHA}	d^{-1}	3.0	6.0
Rate constant for storage of poly-P	q_{PP}	d^{-1}	1.5	4.5
Poly-P accumulating biomass lysis rate	b_{PAO}		0.2	0.13
VFAs half saturation coefficient for storage of poly-P	$K_{lf,P}$	g COD m^{-3}	4.0	1.0
Inhibition coefficient for poly-P storage	K_{IPP}	g P g^{-1} COD	0.02	0.04
PHA half saturation coefficient for storage of poly-P and PAO growth	K_{PHA}	g COD g^{-1} COD	0.01	0.08
Ammonium (as a substrate) half saturation coefficient	$K_{NH4,P}$	g N m^{-3}	0.05	0.01
Phosphate (as a nutrient) half saturation coefficient	$K_{PO4,P}$	g P m^{-3}	0.01	0.001
Hydrolysis (X_S):				
Hydrolysis rate	k_h	d^{-1}	3.0	2.5
Anaerobic hydrolysis reduction factor	n_{fe}	-	0.4	0.1
Temperature coefficients: $\tau = 1.072$ for μ_H, q_{PHA}, q_{PP}, b_{PAO}; $\tau = 1.111$ for μ_A; $\tau = 1.042$ for k_h				

Table 6. Characteristics of the influent fractions and stoichiometric ratios

Fraction	Symbol	Value	Ratio	Unit	Value
sCOD/COD	i_{vt}	0.263	XCOD/VSS	g COD g^{-1} VSS	1.95
BOD5/BODultimate ratio	f_{bod}	0.68	VSS/TSS	g VSS g^{-1} TSS	0.80
Inert fraction of sCOD	f_{rsi}	0.075	BOD/COD	g O_2 g^{-1} COD	0.536
VFA fraction of sCOD	f_{rslf}	0.5	COD/TKN	g COD g^{-1} N	12.7
Substrate fraction of particulate COD	f_{rxs}	0.74	COD/TP	g COD g^{-1} P	86.6
Inert fraction of soluble TKN	f_{rsni}	0.05	NH4/TKN	-	0.549

RESULTS AND DISCUSSION

Model calibration and validation

Sample results of the activated sludge model calibration for the studied plant are shown in Figure 3 and Figure 4. The results present a comparison between measured data and model predictions of the selected contaminants in two-phase (anaerobic/aerobic) batch tests for the summer period and winter period, respectively.

Figure 3a and Figure 4a present results of the tests for PO_4 release and uptake rates under anaerobic/aerobic conditions while Figure 3b and Figure 4b present the results under anoxic conditions (in the second phase). The model predictions for the process rates (continuous line) and the measured data (points) are consistent. The coefficients of determination R^2 are presented in the figures respectively for each process.

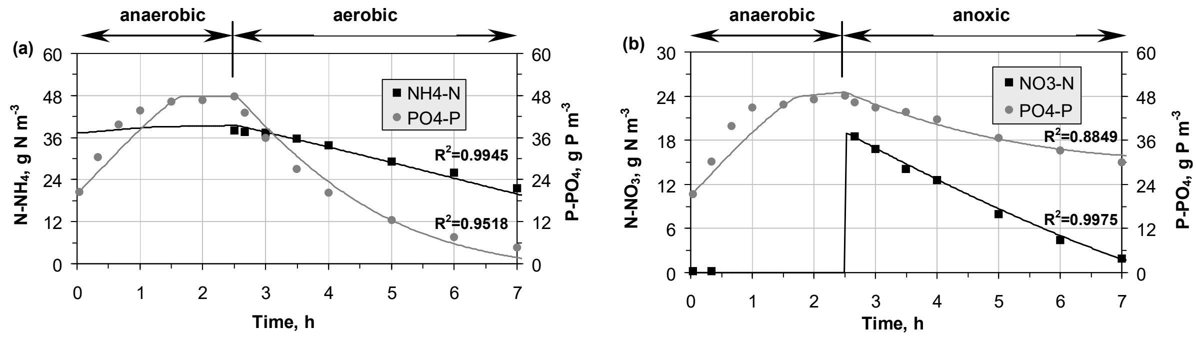

Figure 3. Measured data (points) vs. model predictions (continuous lines) in two-phase batch tests in the summer period: (a) NH_4-N and PO_4-P under anaerobic/aerobic conditions, (b) NO_3-N and PO_4-P under anaerobic/anoxic conditions

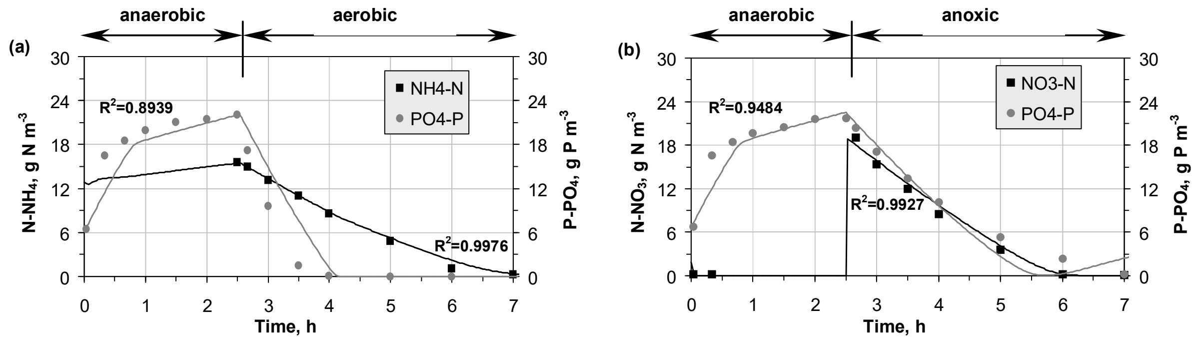

Figure 4. Measured data (points) vs. model predictions (continuous lines) in the two-phase batch tests in the winter period: (a) NH_4-N and PO_4-P under anaerobic/aerobic conditions, (b) NO_3-N and PO_4-P under anoxic conditions

The measurements carried out in the studied plant during the full-scale 4-day campaign were used for the model validation (Figure 5). The diagrams present wastewater flow rate and temperature input data, NO_3 in the aerobic reactor effluent, PO_4 in the anoxic and aerobic zone effluents, and NH_4 in the anoxic and aerobic zone effluents (measured data and model predictions). The model predictions correspond with the measured data. The coefficients of determination R^2 are presented in Figure 5.

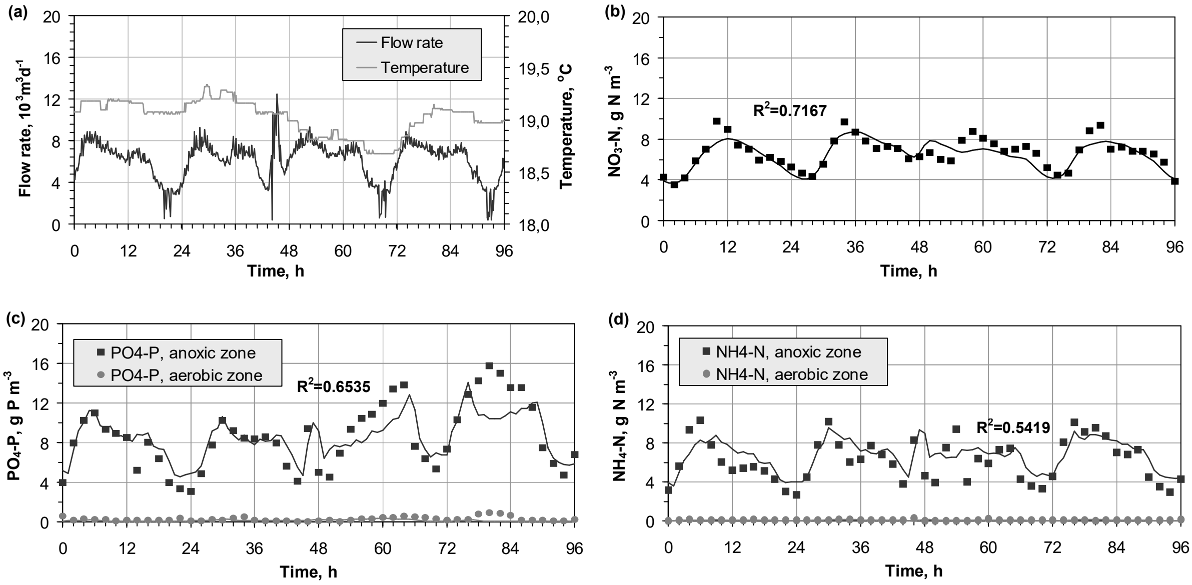

Figure 5. Measured data (points) vs. model predictions (continuous lines) for the 4-day measurement campaign at the studied plant (summer period): (a) input flow rate and temperature, (b) NO_3-N in the aerobic reactor effluent, (c) PO_4-P in the anoxic and aerobic zones effluent, (d) NH_4-N in the anoxic and aerobic zones effluent

Enhanced suspended solids removal

In the analyzed samples of raw wastewater, the COD concentration ranged from 1158 to 2218 gCOD m^{-3}, including 38.5% of the soluble fraction. The average concentrations of nitrogen and phosphorus were respectively 82.9 (±10.4) g N m^{-3} and 16.4 (±2.9) g P m^{-3}. The average TSS concentration was 558 g m^{-3}, including 71% of the VSS fraction. After 2-hour sedimentation (without chemicals addition), the average removal efficiency was 43.3% for COD, 18.9 % for TP, 7.7% for TN, and 59% for TSS (66% for VSS).
The efficiency of enhanced TSS removal was dependent on the doses of ferric (III) sulphate $Fe_2(SO_4)_3$ and anionic flocculent A110 (Kemipol Company/Poland). Sample results for the TSS, COD and TP removal efficiency are presented in Figure 6. The TSS removal efficiency reached the maximum value of 78.6% (87.2% for VSS) for the maximum analyzed doses of the reagents. The efficiency of COD removal remained in the range of 50.5-55.8%. A negative influence of $Fe_2(SO_4)_3$ overdosing was observed at low polymer doses. For the doses of 75 g m^{-3} of $Fe_2(SO_4)_3$ and 1 g m^{-3} of the polymer, relatively high values of removal efficiencies were obtained. The efficiency of particulate COD removal reached 95%. The results obtained in the present study were similar to the results reported by De Feo *et al.* (2008).

The efficiency of the enhanced total phosphorus removal was relatively low (19.7-46.8%). The results were affected by poor settleability of small-size suspended phosphorus fraction. The results for $Fe_2(SO_4)_3$ and the anionic polymer were comparable with the results from the former experiments with organic coagulants (Czerwionka *et al*. 2015). However, in that study, high doses and prices of the organic coagulants led to the negative cost balance.

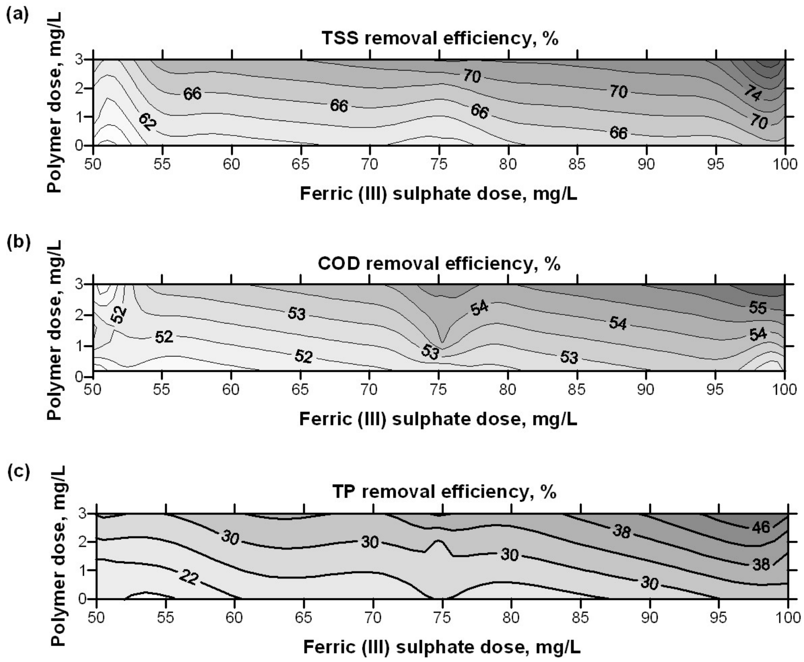

Figure 6. Removal efficiency (%) for TSS (a), COD (b) and TP (c) from raw wastewater for ferric (III) sulphate and anionic polymer A110

Plant-wide model simulations at steady state

Simulations of the plant-wide model at steady-state conditions were carried out after the validation procedure. Wastewater influent characteristics for summer period were used in the study with the set of fractions and stoichiometric ratios listed in Table6. The RAS flow rate (the external circulation) was set at 100% of the influent wastewater flow rate. Table 7 and Table 8 present the simulation results of the plant-wide model vs. the actual results originating from the studied plant, i.e. recorded data for the effluent and biogas generation in anaerobic digestion process respectively. The simulated results are consistent with the measured average values in the studied period.

The results for anaerobic digester were obtained at the temperature of 37°C, VSS loading rate = 1.306 kg VSS m^{-3} d^{-1} and hydraulic retention time (HRT) = 24 d. The values for pH = 7.2, alkalinity = 3433 $gCaCO_3$ m^{-3} and VFA = 57 g m^{-3} were also consistent with the average measured values in the studied period (see: Table 3).

Table 7. Steady-state model predictions vs. actual operational data from the studied plant for wastewater effluent (summer period)

Parameter	TSS	$cBOD_5$	COD	NH_4	NO_2,NO_3	TN	TP
	g m^{-3}	g m^{-3}	g COD m^{-3}	g N m^{-3}	g N m^{-3}	g N m^{-3}	g P m^{-3}
Simulated value at steady state	9.5	3.2	32.0	0.3	7.3	9.7	0.66
Measured value (average)	9.5	3.0	33.1	0.3	7.3	9.7	0.70
Relative error	0%	-6.6%	+3.3%	0%	0%	0%	+5.7%

Table 8. Steady-state model predictions vs. actual operational data from the studied plant for anaerobic digestion (summer period)

Parameter	Unit	Simulated value at steady state	Measured value (average)	Relative error
Raw (primary) sludge mass flow	Mg d^{-1}	4.71	4.54	-3.7%
Waste activated sludge mass flow	Mg d^{-1}	5.40	5.40	±0%
Biogas production	m^3 d^{-1}	3369	3354	-0.4%
Biogas yield per VSS delivered	m^3 kg^{-1} VSS	0.45	0.44	-2.3%
Biogas yield per VSS destroyed	m^3 kg^{-1} VSS	1.09	1.11	+1.0%
Methane fraction of biogas	%	62.0	62.0	±0%
Electric energy production from biogas	kWh d^{-1}	6662	6659	-0.04%
Electric energy consumption for aeration	kWh d^{-1}	6138	6051	-1.4%

Strategies involving the proposed upgrades

Two strategies for upgrading the plant were considered in the present study. The first strategy predicted the impact of increased raw sludge withdrawal as a result of chemically enhanced solids precipitation in the primary clarifier. The second strategy employed the advanced nitrogen removal processes (partial nitrification – anammox) for sidestream treatment. The assumptions made for the analysis focused on maintaining the actual operational conditions including the influent characteristics, RAS recirculation ratio as well as TSS concentration in the primary sludge and WAS.

The TSS removal efficiency in the primary clarifier was a manipulated variable for the first strategy, changing from 30% up to 90%, with a 10% step. The reference state is represented by the value of 40%, while 80% represents the maximum value achieved in the experimental part of the study. In the primary clarifier influent (a mixture of raw wastewater and raw reject water), the summer average value of soluble COD (sCOD)/TKN ratio was 2.86 g COD g^{-1} N at the reference scenario (present conditions) and 2.62 g COD g^{-1} N at 80% TSS removal efficiency. These ratios in the primary clarifier effluent were 3.37 and 3.75, respectively. At the reference scenario, the relationships of BOD:N:P in the primary clarifier influent and effluent were approximately 30:5:1 and 22:4:1, respectively. At the highest predicted TSS removal efficiency (80%), the latter value was approximately 20:6:1 assuming 45% of TP precipitation in the primary clarifier. The fraction of particulate COD (xCOD) in the primary clarifier influent was 0.75 while the fraction in the primary clarifier effluent varied from 0.64 at the reference scenario to 0.37 at the highest TSS removal efficiency. High xCOD/COD ratios observed at the studied plant and other plants with similar influent characteristics are the rationale for removal of higher amounts of primary sludge but a potential carbon deficit for denitrification should be compensated.

The assumptions resulted in changes in the ratio of primary sludge and WAS and the amount of mixed sludge delivered to the anaerobic digester. At the TSS removal efficiency of 80%, the VSS loading rate reached 1.56 kg VSS m^{-3} d^{-1} which was higher by 19% compared to the reference state. Consequently, the HRT decreased to 17.4 days. The simulated VSS destruction in the anaerobic digester and biogas yield per organic matter destroyed increased by 8 and 9%, respectively. In the latter case, the yield reached 1.19 m^3 kg^{-1} VSS.

The efficiency of nitrogen removal from reject water was a manipulated variable in the second strategy, changing from 0% for the reference state up to 90%, with a 10% step. The simulations were carried out for a constant value of the dissolved oxygen concentration = 2 g

O_2 m^{-3} in the aerobic zone of the bioreactor. The observed decrease in the air flow rate did not exceed 5% for the maximum nitrogen removal efficiency. The ratio of nitrogen load in raw reject water to nitrogen load in wastewater influent was only 0.15 for the reference state conditions. On the other hand, a significant change in air supply was observed for the first strategy resulting in the increased reduction (by 25%) for 80% of solids removal efficiency in the primary clarifier.
The electric energy balance was calculated simultaneously for each discrete value of the manipulated variables. The calculations included simulated energy consumption and savings for the activated sludge aeration and potential energy production from biogas. Energy recovery in cogeneration unit was assumed at 32% which followed the actual operational conditions at the studied plant. The biogas calorific value was assessed as 22.3 MJ m^{-3} for 62% CH_4 content. The complementary calculations included the cost balance. The energy costs were calculated for the actual flat rate electric energy price 0.128 $ per kWh. For the chemicals used for precipitation, the following prices were assumed: 0.14 $ kg^{-1} for ferric sulphate and 2.25 $ kg^{-1} for polymer A110. The prices included the cost of transportation. The assumed doses followed the results of the experimental part of the study.
The calculations made for the discrete values of manipulated variables allowed to build grids of the data for technological, energy and cost aspects, respectively. The discrete values were changed into the continuous ones via interpolation made in Surfer 8 software (Golden Software, Inc). Diagrams (maps) created in the software presented a relationship between TSS removal efficiency and nitrogen removal efficiency from reject water as two independent variables, and the selected depended variables subjected to the analysis.

Figure 7 presents results of sensitivity analysis for the strategies involving the proposed upgrade technologies at summer conditions. Figure 7a represents the range of allowable results which are consistent with the Polish regulatory limits for the effluent total nitrogen (10 g N m^{-3}). The maximum acceptable solids removal from primary clarifier was established at approximately 80% provided that simultaneously 90% of the nitrogen load from reject water was removed. The results were also transferred to the other diagrams. The beneficial effect of the first strategy is the increased biogas production and subsequently increased energy production in the cogeneration unit. The biogas production is affected only by the variable representing TSS removal efficiency (Figure 7b). The maximum potential increase in the energy production from methane gas is 56% in comparison with the reference state. Both strategies resulted in the decreased energy consumption for aeration (up to 36%). The energy savings for aeration are strongly affected by the amount of TSS removed from the raw wastewater (Figure 7c).

The cost balance is also strongly affected by the enhanced solids precipitation in the primary clarifier (Figure 7d). Positive results can be obtained in terms of the relationship between the energy price and coagulants/flocculants prices and doses. The reduced energy for aeration as a result of reject water treatment was less significant. For the present level of energy recovery in cogeneration and of energy price for the studied plant, a choice of chemicals for solids precipitation is the main factor determining a positive cost balance.

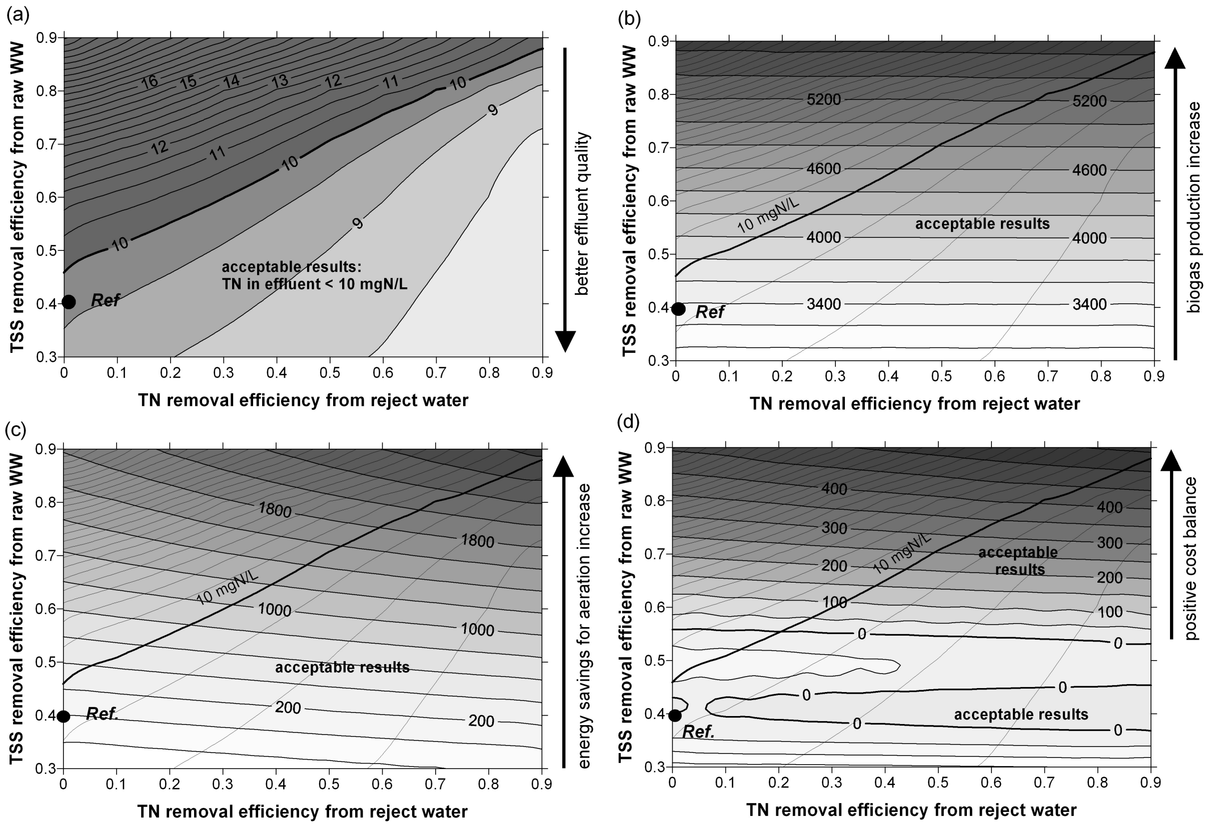

Figure 7. Model predictions of: (a) TN concentration in wastewater (WW) effluent in gN m^{-3}, (b) biogas production in m^3 d^{-1}, (c) energy savings for aeration in kWh d^{-1}, (d) cost balance in USD d^{-1}. "*Ref*" represents the actual value for the studied plant (reference state)

Energy neutrality considerations

The annual average specific electric energy demand was 0.48 kWh per cubic metre of treated wastewater and 0.75 kWh kg^{-1} BOD_5 removed (16 kWh per PE). In a three-year period the specific energy demand changed in the range of 0.42-0.55 kWh m^{-3}. The calculated values indicate that the studied plant has a mean energy consumption in comparison to the other European facilities (0.36 to 0.67 kWh m^{-3} according to Hernández-Sancho *et al*. 2011). The energy balance was affected mostly by aeration system with a share of 53%, followed by pumping with approximately 30%. The specific energy consumption for aeration was 0.56 kWh kg^{-1} BOD_5 removed (12 kWh per PE) while considering only the activated sludge system. The share of renewable energy in total energy consumption at the studied WWTP reached 73%, equivalent to 0.35 kWh m^{-3}.

For the summer period the specific energy demand (0.52 kWh m^{-3}) was higher in comparison with the average annual value and the share of energy from biogas was only 59%. Simulated results of energy neutrality condition for that period are presented in Figure 8. The results indicate that the "neutral point" is located for 75% of TSS removal efficiency and 50% of TN removal efficiency while still maintaining the required level of TN in wastewater effluent. A maximum achievable result was 23% above the energy neutrality level. For 67% of TSS removal efficiency the energy deficit (12%) was similar to the results (19%) given by Gori *et al*. (2013). The WWTP in Strass case showed higher electrical energy production in comparison to energy consumption and 108% of the energy recovery efficiency in 2005 (Kroiss and Cao 2014).

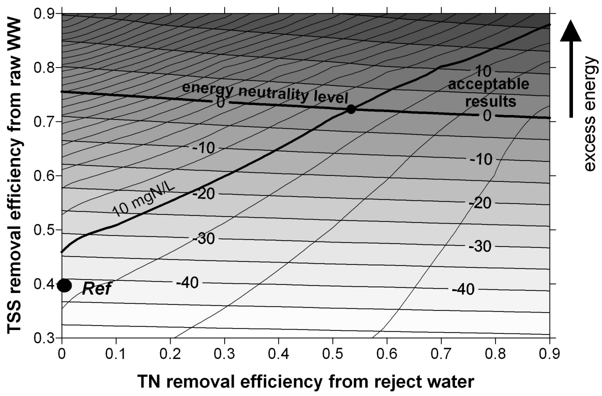

Figure 8. Model predictions under the energy neutrality conditions, in %

CONCLUSIONS

The simulations of the upgraded technologies employing enhanced primary sludge removal and reject water treatment revealed the potential for an increased biogas production in the anaerobic digester up to 56% and reduction in the electric energy demand for aeration up to 36%, while still maintaining the required TN effluent standard. The proposed upgrades improve the energy balance and may lead the studied WWTP from the energy deficit to the energy neutrality. The energy positive area (exceeding the energy neutral point) was found for solids removal efficiency higher than 75% and TN removal efficiency in the sidestream higher than 50%. The operating cost balance depends on the applied coagulants/flocculants and specific electric energy costs. The choice of the coagulant/flocculent was found as the main factor determining a positive cost balance.

A further study should be focused on simulations under dynamic conditions with different influent characteristics and operational parameters. The simulated results should be compared with the measurements in a planned full-scale campaign oriented on enhanced primary sludge removal. The necessary investments and payback time should also be taken into consideration as well as environmental impact and carbon footprint.

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