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IMPROVING THE OPERATION OF THE FULL SCALE WASTEWATER TREATMENT PLANT WITH USE OF A COMPLEX ACTIVATED SLUDGE MODEL

A complex activated sludge model implemented in BioWin software has been implemented to assure its predictability and improve the effectiveness of biological wastewater treatment in the full-scale plant in Poland. The influence of temperature and sludge retention time (SRT) on the quality of the effluent was also studied. The calibration was successfully performed according to the *Good Modelling Practice Unified* (GMP) protocol. Five parameters at a steady state and ten under dynamic conditions were calibrated. It occurred that in the studied wastewater treatment plant SRT should be kept at the low level sufficient to sustain nitrification.

1. INTRODUCTION

For more than 30 years the biological nutrient removal (BNR) technologies of wastewater treatment are widely practiced all over the world. Simultaneously, the development of the mathematical description of biochemical processes responsible for the removal of carbon, nitrogen and phosphorus compounds from wastewater have been observed. As a result, several mathematical models comprehensively describing biological wastewater treatment processes, especially with regard to activated sludge systems have been elaborated. A fundamental meaning in this area had the formulation of the Activated Sludge Model No. 1 (ASM1) by the International Association on

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Water Pollution Research and Control (IAWPRC, the former name of IAWQ and IWA). Although neither biological nor chemical phosphorus removal was incorporated into ASM1, this model provided the matrix notation system and the nomenclature used in further models (*inter alia* ASM2 and ASM2d) [1]. At the same time Barker and Dold [2] formulated a general model for BNR in activated sludge systems. This model was based upon ASM1 with respect to carbonaceous compounds removal, nitrification and denitrification and the model elaborated by Wentzel et al. [3, 4] with respect to biological phosphorus removal. While combining these models certain extensions and modifications were made [2].

The activated sludge models elaborated in last two decades allow the prediction of the effluent composition including the content of carbon, nitrogen and phosphorus compounds. They are very helpful in the optimization studies for the existing wastewater treatment plants (WWTPs) and development of control strategies for the existing or new WWTPs. However, the application of an activated sludge model to a specific plant is a difficult task due to the complexity of these models and large number of data required for the calibration and validation processes.

According to the literature, calibration is performed by changing the values of model parameters and subsequently comparing the modelling results with the field measurements [5, 6]. In the validation process, the model simulation results are compared with an independent set of observations, i.e. the ones not used in the calibration of the real system to verify whether the model describes the system correctly [6]. Several systematic protocols (guidelines) to conduct simulation study, including model calibration, have been elaborated [7]. The International Water Association (IWA) Task Group on Good Modelling Practice (GMP) – guidelines for use of activated sludge models collects knowledge and experience on how to use activated sludge (AS) models in engineering practice [8]. As a result the GMP Unified protocol was elaborated [9]. This protocol was applied here for the purpose of the model calibration. The GMP Unified protocol consists of five major steps: (1) project definition, (2) data collection and reconciliation, (3) plant model set up, (4) calibration and validation, (5) simulation and result interpretation.

The aim of this work was to calibrate a complex activated sludge model to assure its predictability and to improve the effectiveness of biological wastewater treatment in the full-scale plant. In this context, the influence of temperature and sludge retention time (SRT) on effluent quality were also analysed.

2. MODEL CALIBRATION

Below the steps of the GMP Unified Protocol are described. Firstly, the aim of each step is briefly presented and then the step is described with regard to the WWTP studied and the model applied.

STEP 1: PROJECT DEFINITION

The most important part of this step is the formulation of the objectives of the simulation study to be performed. Also all settlements between the WWTP (client) and modeller should be made.

The presented simulation studies are aimed at calibrating a complex activated sludge model implemented in BioWin software for the full-scale plant and analyse the influence of temperature and sludge retention time (SRT) on the effluent quality.

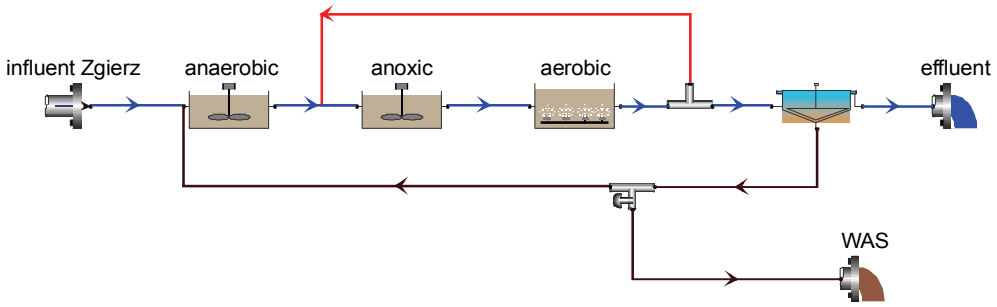

Fig. 1. Schematic flow diagram of the biological reactor at the Zgierz WWTP

The WWTP in the city of Zgierz located in the central Poland was taken into consideration. It treats municipal wastewater from the city and several communes located in the neighbourhood and industrial wastewater originating mainly from small and medium enterprises. The average pollutant load to the plant corresponds to approximately 94 000 PE. The contribution of industrial wastewater is usually in the range from 10% to 15%. The average inflow of wastewater is equal to 11 500 $m^3 \cdot d^{-1}$, whereas the maximum inflow achieves 20 000 $m^3 \cdot d^{-1}$. The biological step consists of one three-zone bioreactor and secondary clarifier run in the modified Phoredox process configuration. The volumes of anaerobic, anoxic and aerobic zone are equal to 857.5 m^3, 3536.4 m^3 and 19 730.6 m^3, respectively. The surface area of the radial secondary clarifier is 1018 m^2 (diameter equals to 36 m) and its total depth is 4.5 m. A schematic flow diagram of the biological step of the Zgierz WWTP is shown in Fig. 1.

STEP 2: DATA COLLECTION AND RECONCILIATION

Step 2 comprises collection of existing (historical) and missing data that must be gathered within a measuring campaign. Separate campaigns were performed to collect the data for the steady state and dynamic calibration. Influent, effluent, sludge excess and recirculation flow rates, pH and temperature were measured on-line in the Zgierz WWTP. Chemical oxygen demand (COD), biochemical oxygen demand (BOD_5), total nitrogen (N_{tot}), total Kjeldahl nitrogen (TKN), nitrate ($N\text{-}NO_3^-$), ammonium ($N\text{-}NH_4^+$), total phosphorus

(P_{tot}), phosphate (PO_4^{3-}), alkalinity, total suspended solids (TSS) were determined with the use of the standard methods [10]. All measurements and analyses were made within the period of data collection for the steady state as well as dynamic calibration.

The data for the steady state calibration and validation were collected in the period of March 2009 and June 2009, respectively. They were collected twice a week at various working days within one month during dry weather conditions. The composite samples were taken from the bioreactor inlet and the plant outlet, and next subjected to aforementioned analyses. All obtained data were statistically elaborated by standard methods. The confidence intervals were calculated by using *t*-Student tests at the significance level of 95%. The averaged values and standard deviations (SD) of the input data introduced into the model are presented in Table 1.

Table 1

Input data for the steady state calibration and validation of a complex activated sludge model implemented in BioWin

Parameters	March 2009		June 2009	
	Mean value	*SD*	Mean value	*SD*
Influent flow rate, $m^3 \cdot d^{-1}$	11 600	1700	11 770	3170
Recirculated activated sludge, $m^3 \cdot d^{-1}$	10 425	1510	10 578	2815
SRT, d	21.0	0.6	22.0	1.0
Temperature, °C	11.8	0.6	17.0	1.0
pH	7.3–7.7*	–	7.4–7.6	–
COD, mg $O_2 \cdot dm^{-3}$	890	233	844	313
TSS, mg TSS·dm^{-3}	298	34	365	109
N_{tot}, mg N·dm^{-3}	59.3	5.1	48.1	19.2
P_{tot}, mg P·dm^{-3}	8.3	1.7	12.6	5.4
Alkalinity, mmol·dm^{-3}	8.4	0.2	8.3	0.2

The data for the dynamic simulations were collected for 48 hours in each measurement campaign. Wastewater samples were collected every two hours from the bioreactor inlet and every 4 hours from the effluent. Their volumes were proportional to wastewater flow. Three independent measurement campaigns were carried out in the period of June, August and September 2009. The results of the campaign performed in June 2009 were used for the model calibration, while the results of the other two campaigns were used in the model validation process.

The correctness of the introduced data was tested using the tools implemented in the software applied, which was described below (step 3).

STEP 3: PLANT MODEL SET-UP

Step 3 includes selection of the simulation platform and the model that will be used in the study. The simulations of biological wastewater treatment processes were

carried out with the help of BioWin v. 3.0 (EnviroSim Associates, Ltd., Canada). BioWin 3.0 uses the integrated activated sludge/anaerobic digestion (AS/AD) model, which is referred to as the BioWin general model. This model is based upon a general model for biological nutrient removal activated sludge systems elaborated by Barker and Dold [2] with a number of modifications and extensions. Due to the fact that in this simulation study only the part of the BioWin integrated AS/AD model referring to activated sludge was used, the model was called here the BioWin AS model. This model includes the following functional categories: 1) growth and decay of ordinary heterotrophic organisms (OHOs), 2) growth and decay of methylotrophs, 3) hydrolysis, adsorption, ammonification and assimilative denitrification, 4) growth and decay of ammonia oxidising biomass (AOB), 5) growth and decay of nitrite oxidizing biomass (NOB), 6) growth and decay of anaerobic ammonia oxidizers (ANAMMOX) and 7) growth and decay of phosphorus accumulating organisms (PAOs) [11].

With regard to data reconciliation, BioWin v. 3.0 checks the correctness of the input data before each steady state or dynamic simulation. It is made for the configuration of the activated sludge system and mass balances for each species. Mass balances are formulated with the assumption that in the bioreactors the liquid phase is completely mixed, the gas phase is completely mixed and the gas hold-up is constant.

In order to check hydrodynamic conditions in the activated sludge system studied, the value of the longitudinal dispersion coefficient (E_L) was estimated with the use of the empirical formulae elaborated by Murphy and Boyko [12]:

$$\frac{E_L}{W^2} = 3.118\left(q_A\right)^{0.346} \tag{1}$$

where q_A is the air flow rate per unit reactor volume ($m^3 \cdot (1000\ m^{-3} \cdot min^{-1})$), W is the reactor width (m). USEPA [13] recommended the use of Eq. (1) as an acceptable approximation of the dispersion coefficient in reactors with both fine and coarse bubble diffused air systems. The value of E_L is equal to zero for true plug flow plants and infinite for true completely mixed plants. It is assumed that plants with a dispersion number (E_L/uL) lower than 0.2 are classified as plug flow, while plants with the dispersion number higher than 4.0 are regarded as completely mixed systems [13]. The letter u in the formulation for the dispersion number stands for the mean displacement velocity ($m \cdot h^{-1}$), while the letter L for the length of the bioreactor (m). The calculated value of E_L/uL in the bioreactor studied was equal to 2.53. According to other authors complete mixing in bioreactors can be assumed when dispersion number is greater than 0.5–4.0 [14] which is in agreement with USEPA [13] recommendation, as well. It indicates that the complete mixing was a good approximation of the hydraulic conditions in the bioreactor of the Zgierz WWTP.

The concentration of dissolved oxygen was set at the level of 3 mg $O_2 \cdot dm^{-3}$ in the aerobic zone.

With regard to settling and separation processes, three models are included in the BioWin AS model, i.e. point separation model, ideal separation model and flux based model [11]. In this study, the ideal separation model was assumed.

Knowledge of the influent composition expressed in COD fractions is one of the most important tasks of the model calibration. In the BioWin AS model five fractions of carbonaceous substrate are distinguished: readily biodegradable including acetate (F_{bs}), acetate (F_{ac}), non-colloidal slowly degradable (F_{xp}), non-biodegradable soluble (F_{us}) and non-biodegradable particulate (F_{up}). Four of these fractions, i.e. F_{bs}, F_{ac}, F_{us} and F_{up} were directly estimated according to the guidelines presented by Roeleveld and van Loosdrecht [15]. These guidelines are based on the physicochemical methods (filtration, flocculation with $Zn(OH)_2$, acetate determination) combined with BOD analysis (measuring BOD in function of time) for determining the fractions of the influent COD. The fraction F_{xp} was estimated as the ratio of non-colloidal slowly degradable COD to the slowly degradable COD. COD fractionation of the influent was made for the composite samples. Thirty independent composite samples were taken from March to September 2009. In Table 2, the results of COD fractionation are presented and compared with the default values of the BioWin AS model. The mean measured values of COD fractions (Table 2) were finally introduced into the model and used in the calibration and validation processes.

Table 2

Characterisation of the influent COD in the BioWin AS model; default and measured values of COD fractions

Fraction	Default value	Mean value measured	Minimum value measured	Maximum value measured
F_{bs}, g COD·g COD_{tot}^{-1}	0.16	0.29	0.24	0.36
F_{ac}, g COD·g $COD_{readily\ biodegradable}^{-1}$	0.15	0.51	0.43	0.60
F_{xp}, g COD·g $COD_{slowly\ biodegradable}^{-1}$	0.75	0.65	0.59	0.68
F_{us}, g COD·g COD_{tot}^{-1}	0.05	0.04	0.025	0.052
F_{up}, g COD·g COD_{tot}^{-1}	0.13	0.23	0.19	0.29

STEP 4: CALIBRATION AND VALIDATION

In step 4, model parameters are adjusted in order to obtain the agreement between the measured data and the results of simulations. Then, the model with the adjusted parameters should be validated using an independent data set.

In Table 3, the values of calibrated parameters for the steady state and dynamic calibrations of the BioWin AS model are gathered and compared with the literature

values or ranges for these parameters. The adjustment of the simulated values of variables characterising the effluent to the measured ones required calibration of five parameters under the steady state conditions and ten parameters under the dynamic conditions. Three (K_S, Y_H, b_H) out of five parameters calibrated in the steady state simulations were associated with growth and decay of OHOs (Table 3). Additionally, AOB decay rate and maximum specific growth rate of PAOs (μ_{maxPAO}) were calibrated. All parameters calibrated under the steady state conditions required also the calibration under the dynamic conditions. Apart from them five additional parameters, mainly associated with the growth and decay of PAOs, had to be calibrated under the dynamic conditions (Table 3).

Table 3

Calibrated values of the BioWin AS model parameters

Parameter	Description	Value			
		Default in BioWin	Calibrated		Literature [1]
			Steady state	Dynamic	
AOB					
$b_{aerob,\,A}$, d^{-1}	aerobic decay rate	0.17	0.23	0.21	0.05–0.15
OHO					
K_S, mg COD·dm^{-3}	COD half saturation constant	5	15	15	4
b_H, d^{-1}	aerobic decay rate	0.62	0.90	0.80	0.2–0.4
Y_H, mg COD·mg COD^{-1}	yield coefficient under aerobic conditions	0.666	0.740	0.700	0.625
$\mu_{max\,H}$, d^{-1}	maximum specific growth rate under aerobic conditions	3.2	–	5.2	3–6
PAO					
$\mu_{max\,PAO}$, d^{-1}	maximum specific growth rate	0.95	0.80	0.80	0.67–1.00
$Y_{PAO,\,aerob}$, mg COD·mg COD^{-1}	yield coefficient under aerobic conditions	0.635	–	0.500	0.625
$Y_{PAO,\,anox}$, mg COD·mg COD^{-1}	yield coefficient under anoxic conditions	0.52	–	0.50	–
$Y_{P/PHA,\,seq}$, mg COD·mg COD^{-1}	amount of PHA stored when 1 mg of acetate or propionate is sequestered	0.889	–	0.959	–
$Y_{P/acetic}$, mg P·mg COD^{-1}	amount of P released for 1 mg of acetate sequestered in the form of PHA	0.49	–	0.54	0.40

The goodness-of-fit can be shown graphically or expressed with the help of an appropriate measure. In order to evaluate the prediction accuracy of the BioWin AS model under the steady state conditions, it was checked, if the simulated value of an output variable was included in the confidence interval estimated for its measured value in the effluent. If it was a case, the simulation was successful because there was no significant statistical difference between the simulated and measured values of the tested variable [6]. In Table 4, the results of steady state simulations were compared with the measured values of the output variables. It occurred that each of the simulated output variables was included in the confidence interval estimated for the respective measured variable in the model calibration under steady state conditions. The same was found in the model validation.

Table 4

Comparison of the measured and simulated values of the output variables under the steady state conditions (March 2009)

Output variable	Measured value	Confidence interval	Simulated value
pH	7.12	0.08	7.18
COD, mg $O_2 \cdot dm^{-3}$	64.0	5.2	59.0
BOD_5, mg $O_2 \cdot dm^{-3}$	6.1	1.2	5.5
N_{tot}, mg $N \cdot dm^{-3}$	13.6	3.1	13.6
$N\text{-}NO_3^-$, mg $N \cdot dm^{-3}$	10.7	1.2	9.8
P_{tot}, mg $P \cdot dm^{-3}$	0.30	0.10	0.35

In the of the dynamic simulations, the goodness-of-fit of the BioWin AS model was checked in a different way. It was made by the calculation of the average relative deviation (ARD) according to the following equation [16, 17]:

$$\mathrm{ARD} = \frac{1}{N}\sum_{i=1}^{N}\frac{|m_i - p_i|}{m_i} \times 100\% \tag{2}$$

where m_i is the measured value of the output variable, p_i – predicted value of the output variable and N – number of the observations.

The ARD values were calculated with respect to the following output variables: pH, COD, BOD_5, N_{tot}, nitrate ($N\text{-}NO_3^-$), ammonium ($N\text{-}NH_4^+$) and P_{tot} in the dynamic simulations after the BioWin AS model calibration. The results are presented in Table 5. The values of ARD varied from 1.6% to 19.2% dependent on the output variable and measurement campaign. This means that the discrepancies between the measured and simulated values of the output variables did not exceed 20%. What is more, excluding two values of ARD, the other ones were lower than 15%. This indicates that the calibration of the BioWin model was performed properly [16, 17].

Table 5

Values of average relative deviation (ARD) calculated for the results of the dynamic simulations

Output variable	ARD [%]		
	June	August	September
pH	4.5	2.2	1.6
COD, mg $O_2 \cdot dm^{-3}$	7.1	14.8	9.3
BOD_5, mg $O_2 \cdot dm^{-3}$	10.2	14.9	13
N_{tot}, mg $N \cdot dm^{-3}$	9.7	14.9	8.9
$N\text{-}NO_3^-$, mg $N \cdot dm^{-3}$	10.2	14.7	11.2
$N\text{-}NH_4^+$, mg $N \cdot dm^{-3}$	8.7	13.2	9.8
P_{tot}, mg $P \cdot dm^{-3}$	19.2	15.1	16.6

To support the presented results of calculations, the measured and simulated values of nitrate are shown in Fig. 2 as an example. They also confirmed the goodness-of-fit of the model after the calibration process. It is claimed that in the dynamic conditions, it is more important to predict the changes of the key output variables at the acceptable level of tolerance than to predict the changes of only one or two output variables ideally [16].

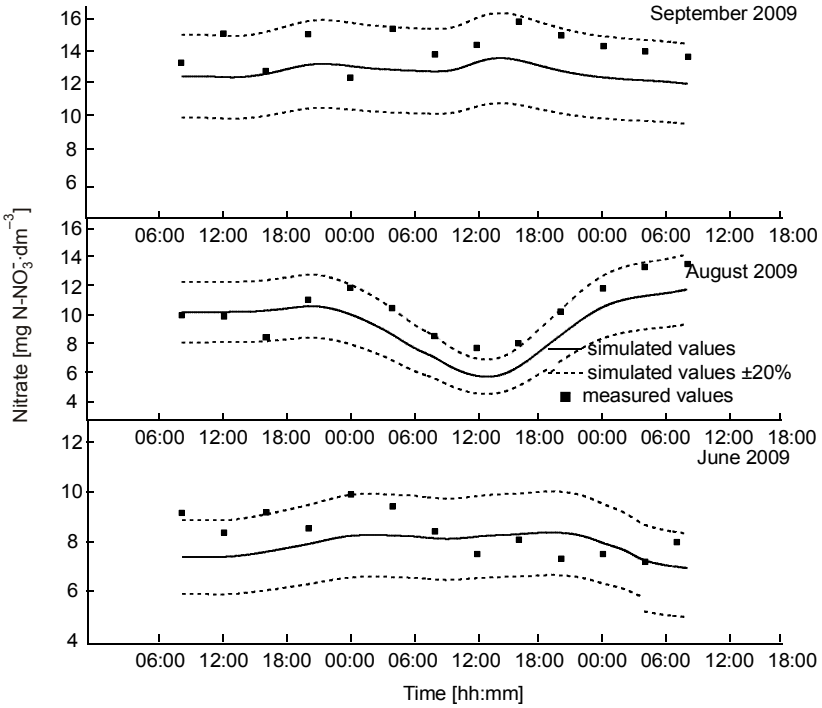

Fig. 2. Measured and simulated values of nitrate ($N\text{-}NO_3^-$) in the effluent under dynamic conditions

At the same time the results presented in Fig. 2 revealed that the problems to keep the concentration of total nitrogen in the effluent below 15 mg $N \cdot dm^{-3}$ incidentally occurred. It was particularly seen in the measurement campaign performed in September, in which the concentration of nitrate varied from 12.6 to 15.9 mg $N \cdot dm^{-3}$. It was also found in the routine monitoring of the effluent quality. The main fraction of total nitrogen in the effluent was nitrate, while the concentration of ammonium was usually below 0.6 mg $N \cdot dm^{-3}$. This indicates that denitrification processes not always ran properly. Therefore, the reconstruction of biological reactor in the Zgierz WWTP was planned. It comprised *inter alia* establishing of two additional zones in the bioreactor: anoxic and aerobic one.

STEP 5: SIMULATION AND RESULT INTERPRETATION

After calibration and validation of any model, various simulations are run to generate data that can be analysed in terms of reaching the aims of a project defined in step 1. In this work, the calibrated BioWin AS model was used to estimate the effect of temperature and SRT on nutrients removal.

Biological nitrogen and phosphorus removal depend on several environmental factors (substrate composition, temperature, dissolved oxygen concentration, pH) and operation parameters (sludge age, sludge loading rate, biomass concentration, temperature) [18]. Out of the operation parameters, SRT is believed to be the one of the most important [18, 19]. In this work, the influence of SRT on the effectiveness of biological nutrients removal was investigated in the range from 10 to 30 d and constant temperature equalled to 17 °C, while the influence of temperature was tested in the range 10–20° C and controlled SRT equalled to 20 d. The selection of the temperature range was made based on the historical data from the Zgierz WWTP. Temperature in the bioreactors of the Zgierz WWTP varied from 12 °C to 19 °C dependent on the weather conditions. At the same time, the operators of the Zgierz WWTP were particularly interested in the optimization of SRT. Thus, a wide range of SRT was tested. The simulations were performed under steady state conditions.

The results of the simulations concerning the effect of temperature on nutrients removal from wastewater are presented in Fig. 3. Generally, temperature in the investigated range had hardly any effect on the effluent quality. It only influenced nitrate concentration in the effluent. It was due to the fact that most nitrifying bacteria are autotrophs sensitive to sudden changes of temperature. The growth rate of nitrifying bacteria increases with temperature between 10 °C and 30 °C, while at higher temperatures between 30 °C and 35 °C the growth rate is constant [20]. Thus, the contribution of nitrate in total nitrogen slightly increased with the increase of temperature. At the same time the total nitrogen remained approximately at the constant level of 10.6 $mg \cdot dm^{-3}$. It was like that because denitrification rate increases exponentially with the increase of temperature from 10 °C to 30 °C.

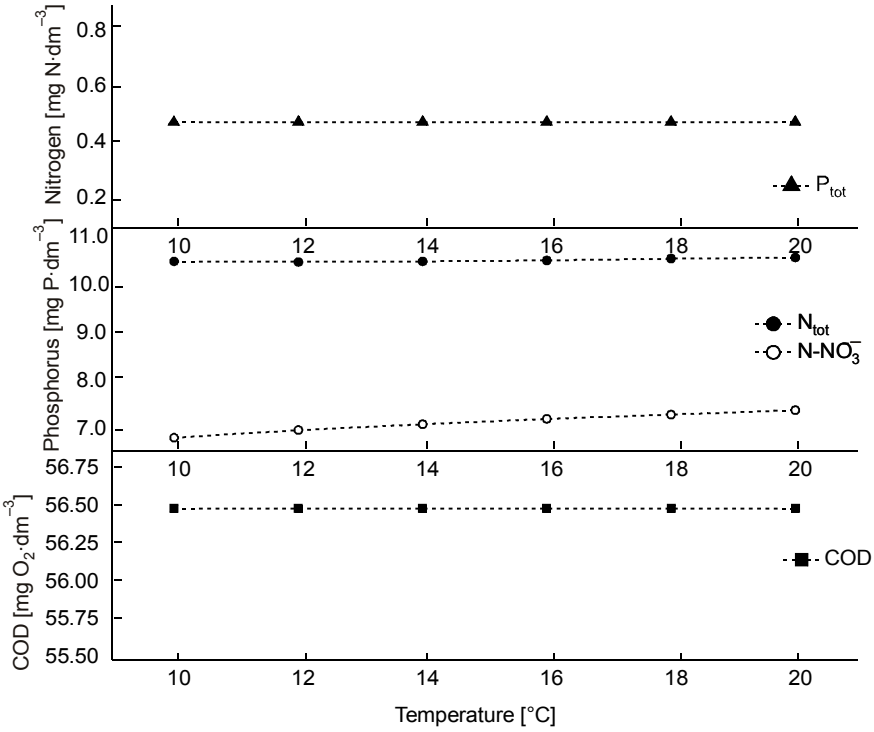

Fig. 3. Effect of temperature on the effluent quality

The changes of SRT exerted the effect on nutrients removal from wastewater. It is well illustrated in Fig. 4. In general, the quality of the effluent declined with the increase of SRT. It is particularly well seen with respect to COD and total phosphorus changes. It was found that COD concentration in the effluent increased with the increase of SRT in agreement with square equation. The correlation coefficient R^2 was equal to 0.994. At the same time the concentration of phosphorus increased linearly with the increase of SRT ($R^2 = 0.983$). This indicates that SRT in the Zgierz WWTP should be kept at relatively low level and should not exceed 20 d. The obtained results revealed that lower values of SRT than those applied in the Zgierz WWTP are sufficient for appropriate removal of nutrients from wastewater.

The above findings are important in the context of the increase of the average pollutant load to the Zgierz WWTP from 94 000 to 108 000 PE in the nearest future and the necessity to fulfil stricter requirements with respect to nitrogen and phosphorus concentration in the effluent. Oleszkiewicz and Barnard [18] reviewed nutrient removal technology in North America and the European Union and formulated the guidelines for achieving the limit of treatment effluent quality for BNR processes. According to these guidelines SRT should be kept as low as possible, just enough to sustain nitrification.

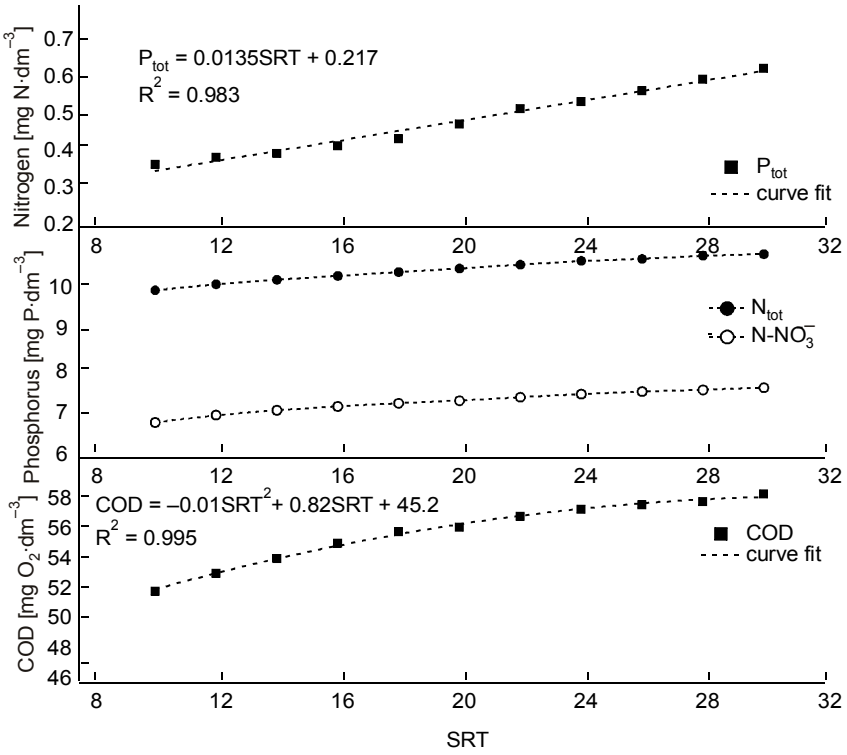

Fig. 4. Effect of SRT on the effluent quality

3. CONCLUSIONS

Each of the simulated output variables was included in the confidence interval estimated for the respective measured variable under the steady state calibration conditions. In the dynamic conditions the discrepancies between the measured and simulated values of the output variables did not exceed 20% on the basis of the average relative degree calculations. It all indicates that the complex activated sludge model implemented in BioWin software was successfully calibrated.

The results of simulations revealed that SRT was more influential than temperature with respect to the effectiveness of nutrients removal from wastewater. Moreover, the increase of SRT deteriorated the quality of the effluent, and thus SRT should be kept at the low level sufficient to sustain nitrification.

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REFERENCES

[1] HENZE M., GUJER W., MINO T., VAN LOOSDRECHT M.C.M., *Activated sludge models ASM1, ASM2, ASM2d and ASM3*, IWA Sci. and Tech., Report No. 9, IWA Publishing, London, 2000.
[2] BARKER P.S., DOLD P.L., *General model for biological nutrient removal activated-sludge systems: model presentation*, Water Environ. Res., 1997, 69 (5), 969.
[3] WENTZEL M.C., DOLD P.L., EKAMA, G.A., MARAIS G.V.R., *Enhanced polyphosphate organism cultures in activated sludge systems. Part III: Kinetic model*, Water SA, 1989, 15, 89.
[4] WENTZEL M.C., DOLD P.L, EKAMA G.A., MARAIS G.V.R., *Enhanced polyphosphate organism cultures in activated sludge systems. Part II. Experimental behaviour*, Water SA, 1989, 15, 71.
[5] STOWA/RIZA, *Smooth modelling in water management. Good modelling practice handbook*, STOWA report 99-05, Dutch Dept. of Public Works, Institute for Inland Water Management and Waste Water Treatment, Lelystad, 1999.
[6] PETERSEN B., GERNAEY K., HENZE M., VANROLLEGHEM P.A., *Calibration of activated sludge models: A critical review of experimental designs*, [in:] S.N. Agathos, W. Reineke (Eds.), *Biotechnology for the environment. Wastewater treatment and modelling. Waste gas handling*, Kluwer Academic Publishers, Dordrecht, 2003.
[7] SIN G., VAN HULLE S.W.H., DE PAUW D.J.W., VAN GRIENSVEN A., VANROLLEGHEM, P.A., *A critical comparison of systematic calibration protocols for activated sludge models: A SWOT analysis*, Water Res., 2005, 39 (12), 2459.
[8] IWA Task Group on Good Modelling, *Practice Guidelines for use of activated sludge models* (online), 2011. Available at https://iwa-gmp-tg.cemagref.fr/test
[9] GILLOT S., OHTSUKI T., RIEGER L., SHAW A., TAKACS I., WINKLER S., *Development of a unified protocol for good modeling practice in activated sludge modelling*, Influents, 2009, 4, 70.
[10] APHA-AWWA-WPCF, Standard methods for the examination of water and wastewater, APHA-AWWA-WPCF, 19th Ed., Washington DC, 1995.
[11] EnviroSim Associates, Ltd., *User Manual for BioWin 3.0*, EnviroSim Associates, Ltd., Hamilton, Canada, 2008.
[12] MURPHY K.L., BOYKO B.I., *Longitudinal mixing in spiral flow aeration tanks*, J. San. Eng. ASCE, 1970, 96 (2), 211.
[13] USEPA, *Manual Nitrogen control*, EPA/625/R-93/010, U.S. EPA, Washington, DC, 1993.
[14] ZIMA P., MAKINIA J., SWINARSKI M., CZERWIONKA K., *Effects of different hydraulic models on predicting longitudinal profiles of reactive pollutants in activated sludge reactors*, Water Sci. Technol., 2008, 58 (3), 555.
[15] ROELEVELD P.J., VAN LOOSDRECHT M.C.M., *Experience with guidelines for wastewater characterisation in The Netherlands*, Water Sci. Technol., 2002, 45 (6), 77.
[16] MAKINIA J., *Mathematical modelling and computer simulation of activated sludge systems*, IWA Publishing, London, 2010.
[17] MAKINIA J., ROSENWINKEL K.H., SPERING V., *Comparison of two model concepts for simulation of nitrogen removal at a full-scale biological nutrient removal pilot plant*, J. Environ. Eng. ASCE, 2006, 132 (4), 476.
[18] OLESZKIEWICZ J.A., BARNARD J.L., *Nutrient removal technology in north america and the European Union: A Review*, Water Qual. J. Res. Canada, 2006, 41 (4), 449.
[19] BARNARD J.L., ABRAHAM K., K*ey features of successful BNR operation*, Water Sci. Technol., 2006, 53 (11), 1.
[20] HENZE M., HARREMOËS P., JANSEN J., ARVIN E., *Wastewater treatment. Biological and chemical processes*, Springer-Verlag, Berlin, 2002.