Cost-efficient Phosphorus removal in rural WWTPs

M. Cramer*, T. Koegst * and J. Tränckner *

* Department of Agriculture and Environmental Engineering, Institute for Water Management, University of Rostock, Satower Str. 48, 18059 Rostock, Germany
(E-mail: *Michael.Cramer@uni-rostock.de; thilo.koegst@uni-rostock.de; Jens.Taenckner@uni-rostock.de)*

Abstract
Phosphorus removal in wastewater treatment plants (WWTPs) is an essential step to mitigate eutrophication in water bodies. In Germany, small WWTP in rural areas have no limit value for phosphorus (P) removal, but can in specific cases or because of accumulative effects be highly relevant for P-loads in surface water bodies. Conventional precipitation is generally feasible but in most cases hardly cost-effective for rural WWTPs, hence, individual methods are needed. This study shows potentials and cost-efficient methods for rural WWTP in comparison with chemical precipitation. Individually customized, they can have a huge potential for P-removal at a small CAPEX and OPEX. Before demanding costly conventional P-removal with fixed effluent standards, a prioritization of most promising plants with regard to the achieved cost-effect relationship is proposed.

Keywords
Phosphorus removal, Wastewater Treatment Plant (WWTP), rural areas, enhanced biological P-removal (Bio-P)

INTRODUCTION

It is generally agreed that increased Phosphorus fractions (P) in the aquatic environment lead to eutrophication of water bodies and hence, should be eliminated in wastewater by treatment plants (WWTPs) (Haandel and Lubbe, 2012). Within the EU, discharge limits on P-emission are defined only for large WWTPs. In Germany this applies for WWTPs of size class 4 and 5 (> 10’000 population equivalents (PE)). However, in rural areas, as common in the state Mecklenburg-Vorpommern (MV), most of WWTPs have a design capacity below 10’000 PE. The effluent P-load of such WWTPs in MV sum up to more than 60% of total emitted P-load (Tränckner et al., 2016). According to Tränckner et al. (2016), P-elimination is strongly dependent on the treatment technology. But, the WWTPs of the same technology show a wide range in P-removal efficiency, that leads to a great individual optimization potential.

The central objective of this paper is the identification of Phosphorus reduction potential of small WWTPs in MV and cost-efficient solutions that are technically feasible in practice. The overall objective has been divided into following sub-objectives:

1. Quantification of P-emissions from small WWTPs in MV
2. Model based identification of optimization potential regarding operational methods
3. Cost estimation for P-precipitation and biological P-storage

Firstly, the relevance of rural WWTPs to ambient water bodies according to P-emission is identified. In agreement with concerned environmental authorities, 19 WWTPs have been selected and analyzed for detailed technologic investigations due to high total P-load in effluent or high ratio of P-emission to ambient receiving water quality.

Secondly, a model to estimate the potential of biological P-removal has been developed. For plants using activated sludge technology, a technological mass-flow model has to be developed for an adequate estimation of P-incorporation as function of wastewater composition, sludge retention time and further parameters to predict the optimization potential. Since conventional methods such as chemical precipitation are not viable for small WWTPs due to high costs and difficulties in manageability in rural areas, the optimization potential of the model is only based on biological and operational methods.

Thirdly, conventional methods of P removal are compared with an enhanced biological P removal (Bio-P) by conditioning the microorganism in those plants, which are capable of integrating such technic with a simple modification of the process. CAPEX (Capital Expenditure) and OPEX (Operational Expenditure) were comparatively calculated to identify cost-efficient solutions for rural WWTPs. Furthermore, a general guideline has been developed to identify and to transfer the findings to relevant “hot-spots” as a step forward to an improved environmental. For evaluating the priority of relevant plants, three evaluation criteria were selected, which combine the emission and ambient water based approach of the EU Water Framework Directive (WFD) with social impact (Cramer et al., 2016).

MATERIALS AND METHODS

Data collection

The processed data have been supplied by the environmental state database hosted by the environmental department MV. Particular technical data, including quality measurements of concerned WWTP’s, have been obtained in cooperation with WWTP operators. To verify the optimization assumptions, selected plants are considered and determined in detail.

Mathematical model

Chemical P-elimination

Chemical elimination is based on precipitation with trivalent metals. The required P-elimination load $X_{P,AE}$ is calculated for existing WWTPs by the difference of effluent P-load and target value. The demand of chemical precipitants (CP) is calculated as:

$$X_{P,AE} = X_{P,influent} - X_{P,effluent} \tag{1}$$

The demand of chemical precipitants (CP) Q_{CP} is calculated by:

$$Q_{CP} = \frac{M_{CP}}{M_P}\beta \cdot B_{P,AE} \cdot \frac{M_{CP} + \sum_{i=1}^{n} x_i \cdot M_{AS,i}}{M_{CP} \cdot c_{solution} \cdot \rho_{solution}} \cdot Q_{influent} \tag{2}$$

where M is the molar mass of CP, P and attendant solution (AS), respectively, β is the excess factor, Q the volume, c the concentration and ρ is the density of the precipitant solution.

Biological P-elimination in activated sludge systems

In activated sludge systems (ASS) (E.g. oxidations ditches and SBR plants), the potential of P-incorporation into biomass is a function of COD in raw water, sludge retention time (SRT) and other parameters. In Germany, design of ASS is based on DWA-A 131 (DWA, 2015). This manual can be used to balance biomass production based on COD. Thereby, P removal is simply assessed with a fixed factor of COD. In this work we modified the calculation of P incorporation by connecting it directly to the biomass production.

Total sludge production is calculated according to the scheme in Figure 1. The particulate COD fraction (CODp) is estimated from the volatile suspended solids (VSS) at the inflow. If this value is unknown, it can be estimated from total suspended solids (TSS) with an estimate for the mineral fraction. Dissolved inert COD (S_i) that does not contribute to biomass production is estimated to be 5% of the COD inflow. A good guess is also the effluent COD minus the COD of the TSS at the effluent. Inert particulate COD (X_i) contributes directly to sludge production. The degradable COD is transformed into biomass, reduced by decay. The decay products contain an inert ratio $a_{Xi,decay}$. Sludge production (SP) comprises accordingly X_i, X_H and $X_{i,decay}$. With stoichiometric factors, the different COD fractions are transferred into TSS. For the biomass fractions, the mineral constituents must also be considered. According to Ramdani et al., (2010), about 92% of biomass are organic. Since biomass production is a function of SRT and SRT depends on sludge production, for a given plant an iterative procedure, starting with an estimate of SRT, is required. Suggested values for COD fractioning, decay rate and transferring COD into TSS are given in Table 1.

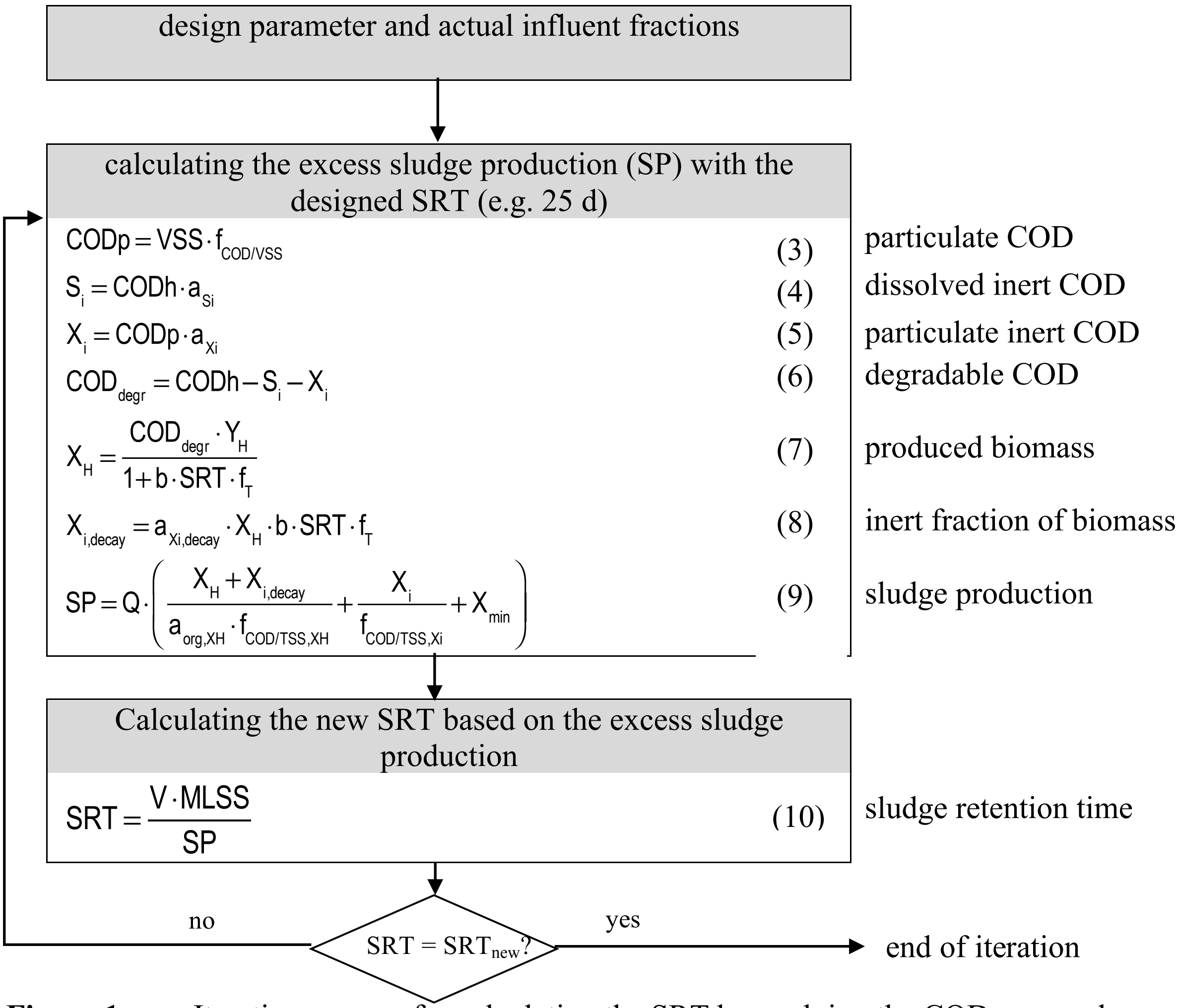

Figure 1. Iteration process for calculating the SRT by applying the COD approach

For calculating P removal, only the produced biomass X_H is relevant. Ordinary heterotrophic organism incorporate about 2.5 % P-fraction in relation to total biomass. This value can be enhanced up to 5% by activated sludge conditioning through enhanced P-elimination (Bio-P)

(Haandel and Lubbe, 2012). Depending on actual condition, biologic P incorporation is calculated with Eq. 11.

$$X_{P,elimniated} = \frac{X_H}{f_{COD/TSS,XH}} \cdot f_{P,BM} \tag{11}$$

$f_{P,BM} = 0.025$ for plant without enhanced biological P-elimination

$f_{P,BM} = 0.05$ for plant with enhanced biological P-elimination

Table 1. Parameter estimation for activated sludge systems in urban wastewater

Parameter	Symbol	Value	Literature
Yield	Y_H	0.67 $g_{COD}\ g_{COD,deg}^{-1}$	(DWA, 2015)
Decay rate	b	0.17 d^{-1}	(DWA, 2015)
Design temperature	T	10°C	
Inert dissolved COD-fraction	a_{Si}	5% of COD_h	(DWA, 2015)
Inert particulate COD-fraction	a_{Xi}	25% of COD_p	(DWA, 2015)
Mineral residue of particular fraction	$a_{Xi,decay}$	20%	(DWA, 2015)
Organic matter of biomass	$a_{org,XH}$	92%	(Ramdani et al., 2010)
COD to TSS ratio	$f_{COD/TSS,XH}$	1.45 $g_{COD}\ g_{TSS}^{-1}$	(DWA, 2015)
Particulate COD-fraction	$f_{COD/TSS,Xi}$	1.6 $g_{COD}\ g_{TSS}^{-1}$	(DWA, 2015)
Chemical oxygen demand	COD	120 $g_{COD}\ PE^{-1}d^{-1}$	(DWA, 2015)
Mixed liquor suspended solid	MLSS	50 g $PE^{-1}d^{-1}$	(Friedrich, 2014)
Mineral dry matter content	X_{min}	20 g $PE^{-1}d^{-1}$	(Imhoff, 2007)
Temperature factor	f_T	1.072^{T-15}	(DWA, 2015)
Organic biomass fraction	$a_{org,XH}$	92%	(DWA, 2015)
Inert ratio	$a_{Xi,decay}$	20%	(DWA, 2015)
	$f_{COD/VSS}$		(DWA, 2015)

In most cases, integration of a Bio-P requires only minor modifications on existing WWTP configuration and operation. For this, a switch of anaerobic and aerobic phases is required, whereby the time interval of the anaerobic phase should be in the range of 1 – 2 hours. Excess sludge has to be removed at the end of the aerobic phase, e.g. at the stage with maximum P uptake. Detailed information on basics and application of enhanced Bio-P are given in Haandel and Lubbe, (2012).

Biologic P-elimination in biofilm systems and near natural systems

In contrast to ASS, P removal in biofilm and near natural systems are much less predictable and largely governed by specific operational conditions. For those systems, a reliable mathematic method to assess the actual P removal based on design parameters is yet not available. In this work, achieved P removal efficiencies from different systems has been calculated based on survey data of environmental authorities from the year 2014 (average value from monthly taken samples). The selected WWTP are depicted in Figure 2.

RESULTS AND DISCUSSION

Quantification of P-emissions from small WWTPs in MV

In the federal state of MV, characterized by rural land use, activated sludge systems such as SBR plants and near-natural systems such as sewage ponds are commonly applied for wastewater treatment (Figure 2). In case of MV, Figure 3 shows that activated sludge systems (ASS) are by far more efficient regarding P-elimination than sewage ponds (SP) and biofilm systems (BS). WWTPs with an integrated chemical precipitation are excluded. It is further more apparent that, WWTPs in MV show a huge range of P-removal efficiency. Hence, this leads to a need of further investigations of WWTPs especially with poor efficiency to identify the individual problems of this plants. Furthermore, it can be seen that activated sludge system and sewage ponds are the dominating treating methods in MV due to the high P-load percentage.

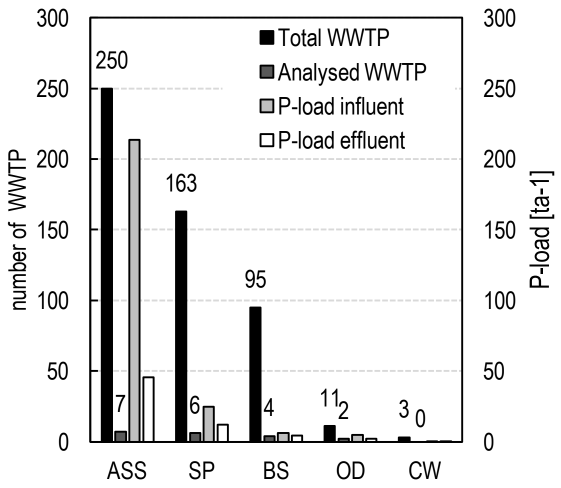

Figure 2. Overview of the treatment technology of WWTP in MV with less than 10’000 PE from the year 2014

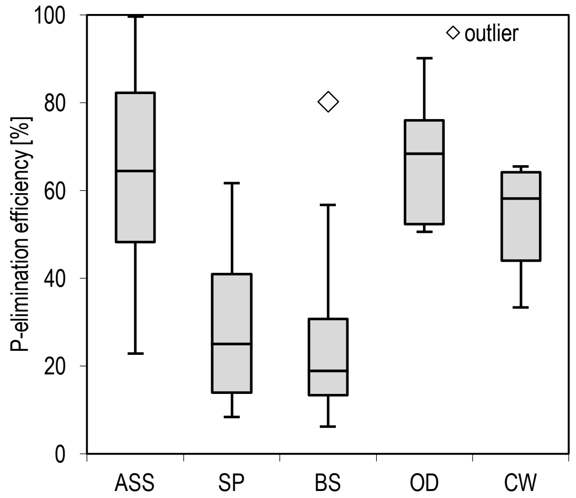

Figure 3. Overview of P-load and efficiency of WWTP in MV with less than 10’000 PE from the year 2014

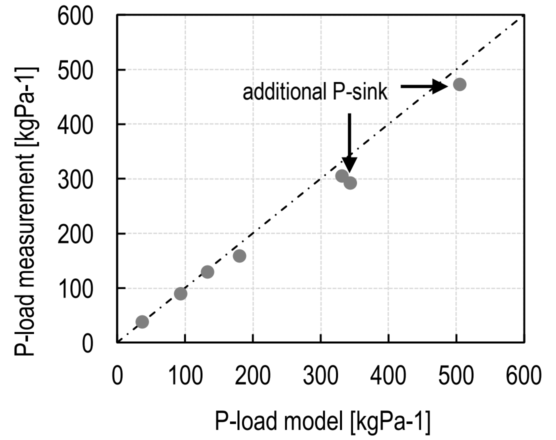

Figure 4. P-effluent-load for model verification based on 7 activated sludge systems

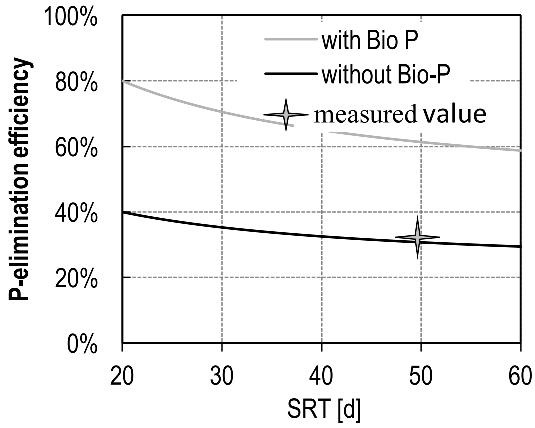

Figure 5. P-elimination efficiency in dependence of SRT of one selected WWTP

Model based identification of optimization potential regarding operational methods

Model verification

For verification of the model described above (Figure 1), calculated P removal of 7 WWTP has been compared to measured data (Figure 4). With the exception of two plants the deviations are less than 5%. These both plants are equipped with a sewage pond, which obviously have a small additional removal effect. Therefore, the model fits well to activated sludge systems and is highly suitable for prediction of biological P-removal.

The developed model sets the basis for identification of optimization potentials as exemplarily illustrated in Figure 5 for one selected WWTP. Currently, without enhanced biological P-elimination (Bio-P) and a calculated SRT of 50 d, it provides a P removal efficiency of 32 % (=10,6 mg/L at the effluent). Simply a reduction of SRT of 25 d could improve the P-removal to 37% (=9.6 mg/L). Additional enhanced Bio-P could achieve a P-removal of 75 % (=3.9 mg/L).

SRT reduction

Based on the investigated plants, huge SRT in rural WWTP can be indicated as a key factor of low P-removal efficiency. The high SRT is a result of a combination of generous design, shrinking population and operational conditions. High SRT of above 100 days were not uncommon and lead to substantially reduced production of highly stabilized excess sludge. This is dearly bought by an enhanced aeration demand and a low P-removal.

In general, a SRT in rural WWTP of about 25 d is recommended to ensure an aerobic stabilization of the sludge. If the sludge is disposed to a central anaerobic digestion (as meanwhile often done), the SRT can even be reduced according to the desired wastewater treatment processes. Due to the high toxicity of NH_3, generally a nitrification is intended. In combination with denitrification a total sludge age of 12 - 15 d is sufficient. If in exceptional cases a nitrification is not required, even lower SRT can be taken into consideration (Figure 6). However, the increased sludge production and its lower stabilization must be regarded in the context of the storage and disposal strategy and the related logistic effort and possible side effects e.g. odor. In each case it is mandatory to store excess sludge and primary sludge separately to avoid P release from the stored biomass.

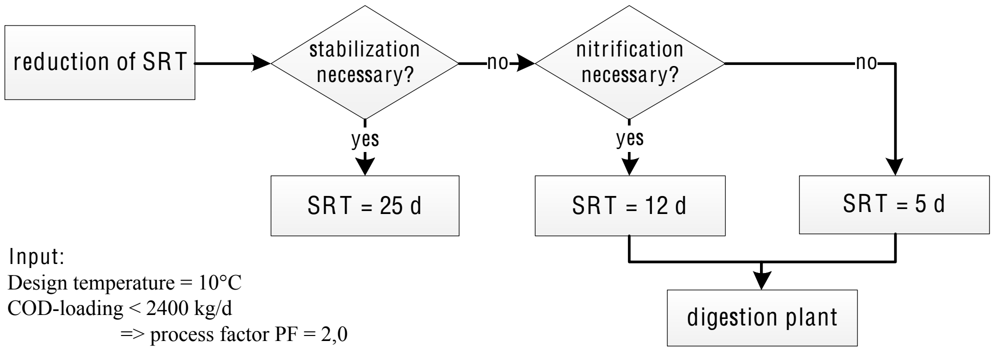

Figure 6. Determination of the minimum possible sludge retention time

Biofilm-systems

From an economic point of view, the key advantages of biofilm-systems are the low excess sludge production and low energy consumption. Logically, the invers conclusion is a poor P-elimination efficiency. Moreover, this fact is reinforced by the mutual storage of excess sludge from the final clarification with the primary sludge, which is commonly done in trickling filter. Mixing the activated sludge with well biodegradable organic matter enhances the biological degradation process and leads to an uncontrolled cold fermentation. This causes a dissolution of P-fractions stored in the biomass and returns it with the backwater. Hence, as a cost-efficient solution, a separate excess sludge tank is recommended. Practical experience show an optimization potential of up to 30%, according to two comparable plants with and without separate excess sludge tanks. This potential of separate storage should be in analyzed in detail in further investigations for a deeper understanding. Other optimizations are difficult to integrate into biofilm-systems. In particular integration of Bio-P is not possible. Figure 7 shows a common and an optimized trickling filter. The same applies for packed bed reactors.

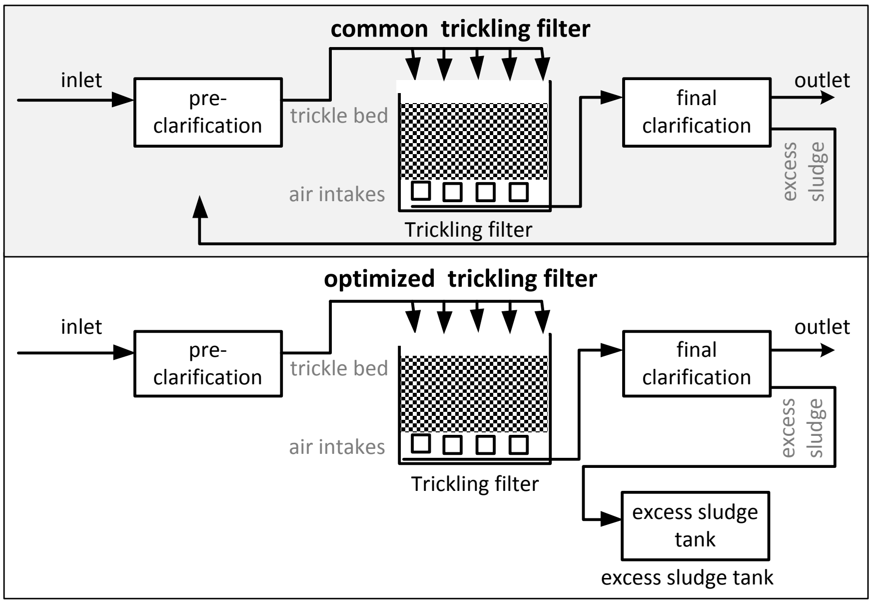

Figure 7. Schematic treatment technology of trickling filter

Near to nature processes

The options for operational optimization of near to nature processes such as sewage ponds are marginal or nonexistent. Even in the cases of largely decreasing loads, alternative systems for binding P-fractions in biomass such as swimming plants systems with waterweed are not economic viable due to the need of frequent harvesting and disposal of the plants. For instance, the P binding rate of duckweed Lemna has experimentally assessed to lie between 78 and 754 $kg_P\ ha^{-1}a^{-1}$ (Bonomo et al., 1995; Körner et al., 2003). Given a P load of 1,5 g $cap^{-1}d^{-1}$) and a desired removal efficiency of 50%, the required additional pond surface is 8 to 75 m^2cap^{-1}. In comparison, conventional pond systems are designed in Germany with 3 m^2CAP^{-1} for aerated pond and 12 m^2cap^{-1} for non-aerated ponds, respectively (DWA-A 201, 2005).

A promising approach for compensating the low excess sludge production in sewage ponds and therefore poor P-elimination efficiency is a modification of one pond to a SBR-pond-reactor. This leads to a great potential due to the enhanced excess sludge production, which is comparable with conventional activated sludge systems and by virtue of the ability of integrating Bio-P. Hence, the developed model can be applied to this modified SBR-pond to estimate the optimization potential. Figure 8 shows a scheme for a technological shift from a conventional pond to a SBR pond. A negative side effect of installing an SBR pond is a decreasing stability of the pond banks due to the variable water height. This applies namely for simple basins without sheeting and enforcement.

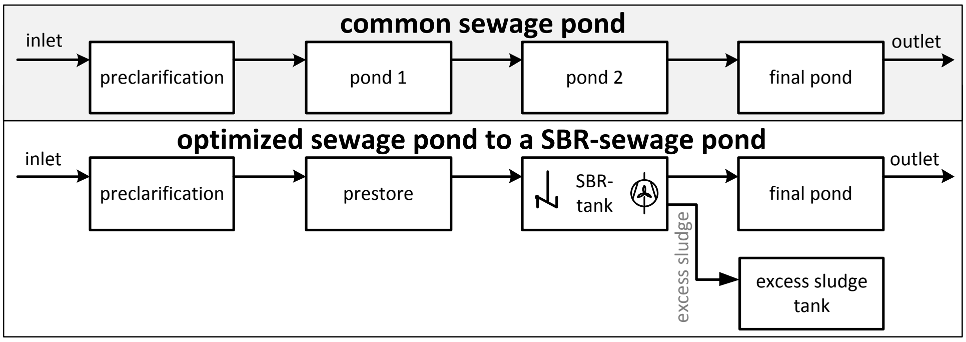

Figure 8. Schematic treatment technology of sewage ponds

Activated-sludge processes (SBR, oxidation ditch, compact system)

All SBR-plants are suitable for integrating a Bio-P. In most of the cases in MV this has been already done. Most of the analyzed plants showed a far to high SRT, which causes a low excess sludge production. Generally, the SRT could be reduced without or with marginal invest costs by reducing the volume of the reactor by lowering the water level and by reducing the activated sludge concentration in the reactor, respectively. However, below a certain concentration activated sludge does not flock well, leading to bad sedimentation behavior. Generally, sludge concentrations above 2 gL^{-1} are not critical.

Typically, an oxidation ditch (Figure 10) is not known for enhanced biological P-elimination caused by the high throughput speed. Investigations on a particular oxidation ditch showed however a kind of an integrated Bio-P with astonishing high P contents in the mixed liquor of 5 - 6.5 $g_P\ g_{VSS}^{-1}$. The plant is aerated by only one mammoth rotor, leading to the assumption of an uncontrolled formation of anaerobic and aerobic zones. However, grab samples along the whole ditch were partly oxygen-free but contained always Nitrate concentrations of at least 10 mgL^{-1}, which would hinder a P release. It can be assumed that the low turbulence in parts of the ditch leads to layering with anaerobic zones near the bottom. These can alternating raise when passing the rotor. In contrast, a plant with two aerators and a high velocity showed a P-fraction of 2.3% of the organic matter, which is a typical value for sludge without Bio-P. Here, no concentration oscillations of P, nitrate and oxygen, respectively, have been observed. The processes in this plant are still under investigation to understand the hydrodynamic and bio-chemical processes better and deduce advices for controlled optimization of the aeration scheme. Therefore, the developed model can be applied to oxidation ditches as well.

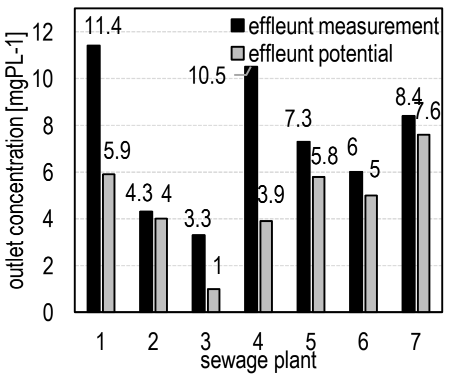

Figure 9. Optimization potential of selected activated sludge systems

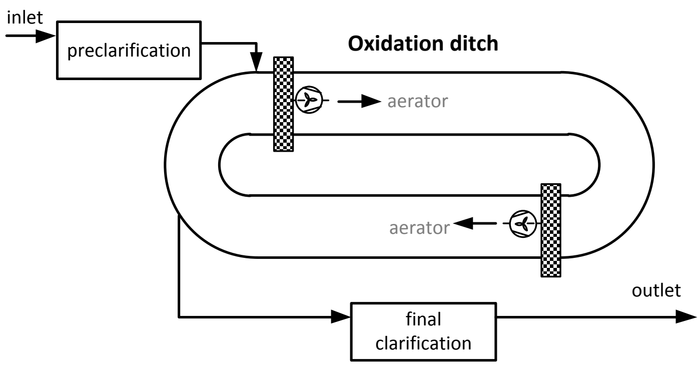

Figure 10. Schematic of the treatment technology of an oxidation ditch

Summarizing, Figure 9 shows the total potential of optimization of selected plants with ASS systems, just by adjusting SRT and introduction of Bio-P where feasible. The plants 1 and 2 are oxidation ditches without and with Bio-P, respectively. Plant 3 and 4 are SBR without Bio-P and the plants 5,6 and 7 are SBR with integrated Bio-P.

Cost estimation for P-precipitation and biological P-storage

The optimization of P-removal by reducing the SRT and/or introduction of enhanced Bio-P removal has technological side effects. The most relevant ones are increased sludge production and reduced oxygen demand by reduced energy demand. In most cases, savings of the energy demand will overcompensate the increased disposal costs.

The conventional alternative is chemical precipitation using Iron or Alum salts. This demands the installation of a dosage system, consumes the precipitation mean and produces additional precipitation sludge. For very small plants with low consumption of precipitants, the dosage stations can be simple, consisting of a dosage pump directly fixed on vessel. Those systems are available at end-of-pipe solutions on the market. For larger plants, container solutions where the precipitant is stored in an IBC container are a pragmatic solution. According operators experience in Germany the investment costs differ between 2000 € for the simple system to 20.000 € for the container solution, both at a depreciation period of 15 years. For logistic reasons (interval of refilling the vessel) the simple solution can be used roughly up to a plant size of 800 cap. The costs for the precipitants differ widely depending on the respective product, the demand and logistic questions. They also vary in time according to the market situation. In this case, the costs for precipitants are estimated with 0,80€. The assumed rate of interest is 3.5%. The annual cost are calculated according to the German Working Group on Water Issues of the Federal States and the Federal Government (LAWA, 2005).

Summarizing, an indicative cost comparison of conventional precipitation and operational optimization, introducing an enhanced Bio-P shall be essayed as function of plant size. Figure 11 shows the total annual costs and the per-capita costs. While the per-capita costs are nearly negligible for the enhanced Bio-P, they range between 5 to 6,5 €$cap^{-1}a^{-1}$ when introducing chemical precipitation. The jump at 800 PE is due to the change from the compact dosage station to the container solution. Although the value will change when using different input data, the tendency remains. Hence, before asking for chemical precipitation in small WWTP the operational optimization potential should carefully checked. Unfortunately, Bio-P and other optimization measures are less reliable over time and will therefore not always meet fixed discharge limits. But, since Phosphorous is a non-toxic eutrophication parameter, the accumulated yearly load is more decisive.

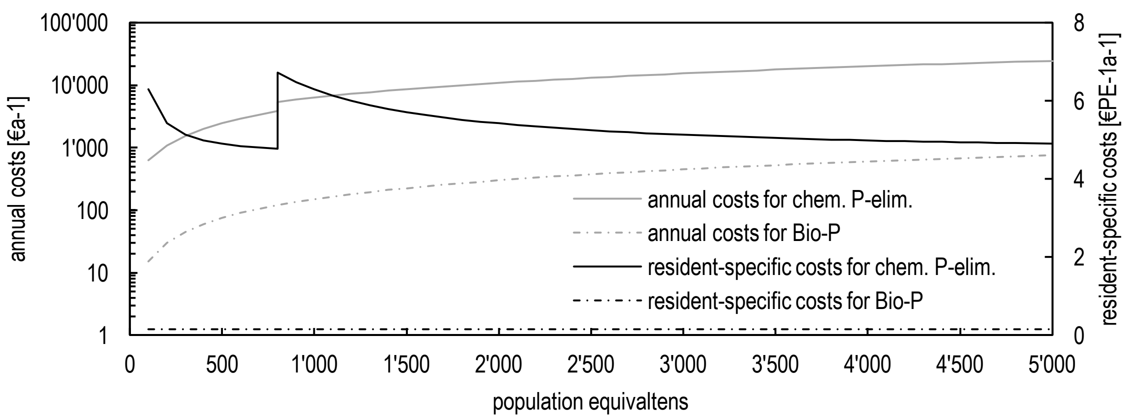

Figure 11. Comparison of the costs of biological and chemical methods, conditions: disposal costs: 300 €t^{-1} excess sludge

CONCLUSION

Based on the study, the following conclusions are drawn:

- Small WWTP have in sum a huge impact on the nature and needed to be more focused in the future to make a step forward to an enhanced environmental
- The developed model fits well to ASS
    - The P-removal as a function of SRT can be precisely estimated
    - The model is suitable to predict the biological P-removal with and without Bio-P
    - The biological optimization potential can be estimated
- Small WWTP in rural areas provide a huge optimization potential
    - Bio-P is not in each ASS integrated and could be done with changing the operational mode
    - A technology switch from sewage pond to SBR-pond improves the P-elimination significant
    - Excess sludge and primary sludge of biofilm systems should be stored separately
- Additional costs of integrating a Bio-P are marginal with respect to a chemical precipitation
    - Biological methods for P-removal leads to cost-efficient solutions

REFERENCES

Bonomo, L., Pastorelli, G., Zambon, N., 1995. Advantages and Limitations of duckweed-based wastewater treatment systems. Water Science and Technology 35 (5), 239–246.

Cramer, M., Koegst, T., Tränckner, J., 2016. Reduction of P in surface waters by improved P removal in rural WWTP. 2016, 1–10.

DWA-A 131, 2015. Bemessung von einstufigen Belebungsanlagen. ATV-DVWK-Regelwerk 131 : Arbeitsblatt. GFA, Verl. für Abwasser, Abfall und Gewässerschutz, Hennef.

DWA-A 201, 2005. Grundsätze für Bemessung, Bau und Betrieb von Abwasserteichanlagen, August 2005 ed. DWA-Regelwerk A 201. DWA, Hennef.

Friedrich, M., 2014. Rostocker Abwassertagung - Infrastruktur und Energiemanagement - ein Geschwisterpaar der Wasserwirtschaft: Beitrag: Aspkete der Energiewirtschaft bei der Abwasser- und Schlammbehandlung. Universität Rostock, 2014.

Haandel, A.C., Lubbe, J.G.M., 2012. Handbook of biological wastewater treatment: Design and optimisation of activated sludge systems, 2nd ed. ed. IWA Pub., London, England, New York.

Imhoff, K., 2007. Taschenbuch der Stadtentwässerung. Oldenbourg Industrieverlag.

Körner, S., Vermaat, J.E., Veenstra, S., 2003. The Capacity of Duckweed to Treat Wastewater. Journal of Environment Quality 32 (5), 1583.

LAWA, 2005. Leitlinien zur Durchführung dynamischer Kostenvergleichsrechnungen (KVR-Leitlinien). Aufl., Berlin.

Ramdani, A., Dold, P., Déléris, S., Lamarre, D., Gadbois, A., Comeau, Y., 2010. Biodegradation of the endogenous residue of activated sludge. Water Research 44 (7), 2179–2188.

Tränckner, J., Koegst, T., Cramer, M., Gießler, M., Richter, B., Müther, F., 2016. Phosphor-Elimination in Kläranlagen bis 10.000 Einwohnerwerte in Mecklenburg-Vorpommern., 2016, 1–121.