Artemii Osipchuk

USE OF ACTIVATED SLUDGE PROCESS MODELS.

Application at small-scale wastewater treatment plant in Finland

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Abstract

Mathematical modelling of the Activated Sludge Process (ASP) has gained close attention as a valuable tool in the wastewater treatment during the last decades. Modelling is widely applied at large-scale Wastewater Treatment Plants (WWTPs) for design, optimisation, research and control using simulation software based on standardised mathematical models such as the Activated Sludge Model (ASM) family. However, the application of modelling at small-scale WWTPs is uncertain due to a procedure complexity and local use cases.

Lapinjärvi is a small Finnish municipality in the Uusimaa region with three small-scale operating WWTPs. The topic of the study is the largest one, the Kirkonkylä WWTP. The current treatment process does not achieve favourable cleaning results due to ageing equipment and an outdated operation approach. The municipality needs a new method for the process control of operating WWTPs and planning a new WWTP.

The focus of this study is an evaluation of modelling applicability at a small-scale WWTP. The possibility of WWTP model simulation with data limited to the conventional construction and operational data is considered. The results of the steady-state simulation and possible applications are demonstrated in practice.

In this work, the developed model of the Kirkonkylä WWTP with oxidation ditch configuration of the ASP is described. The model of the plant is built on the base of the Activated Sludge Model No. 1 (ASM1) combined with the Takács model based on the model of Vitasovic for the secondary settling tank. The plant model and simulations are completed in the WEST 2017 SP 1 software environment released by MIKE Powered by DHI.

Keywords

wastewater, treatment, modelling, simulation, activated sludge, oxidation ditch

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Now I feel ready to go out into the world promoting constructive engineering ideas and sustainable development on the way.

Artemii

LIST OF ABBREVIATIONS

ASM1 – Activated Sludge Model No. 1

ASP – Activated Sludge Process

ASR – Activated Sludge Reactor

BOD – Biochemical Oxygen Demand

COD – Chemical Oxygen Demand

CSTR – Continuous Stirred Tank Reactor

DO – Dissolved Oxygen

F/M – Food-to-Microorganism Ration

GMP – Good Modelling Practice

HRT – Hydraulic Retention Time

MBR – Membrane Bioreactor

MLSS – Mixed Liquor Suspended Solids

MLVSS – Mixed Volatile Liquor Suspended Solids

RAS – Return Activated Sludge

SBR – Sequencing Batch Reactor

SETTLER – Secondary Settling Tank

SRT – Solid Retention Time

TKN – Total Kjeldahl Nitrogen

TN – Total Nitrogen

TSS – Total Suspended Solids

WAS – Waste Activated Sludge

WSP – Wastewater Stabilisation Pond

WWTP – Wastewater Treatment Plant

CONTENTS

1 INTRODUCTION

Wastewater Treatment Plants (WWTPs) play a critical role in reintroducing contaminants back into nature’s cycle with limited environmental impact. Wastewater treatment is a complex multi-stage process that requires high precision to achieve high-quality outputs. Taking into account guidelines and regulations, e.g. the EU regulation (Council Directive 91/271/EEC) and the Finnish regulation (Government Decree on Urban Waste Water Treatment 888/2006), that require specific effluent quality levels, it becomes necessary to provide a more efficient process and control procedures for wastewater treatment.

Biological treatment is the essential stage in the wastewater treatment process, where biodegradable contaminants are removed. The Activated Sludge Process (ASP) is the most effective method used as the biological treatment stage because of its cost-effectiveness, reliability and flexibility. The operation of the ASP requires enhanced knowledge and a high accuracy as a complex set of parameters should be monitored and adjusted to keep the process smooth.

Mathematical modelling of the ASP, which found its origins in the study by Downing et al. (1964), is used for the optimization of the wastewater treatment leading to a reduction of operation costs and high effluent quality. The ASP modelling becomes an indispensable tool integrated into a regular basis at large-scale WWTPs processing high wastewater flow. In opposite, the use of the ASP modelling is negligible at small-scale WWTPs, where a wastewater flow is comparatively low. Mainly, it is due to difficulties, lack of resources and the absence of a clear understanding of what benefits it can provide for small-scale systems.

1.1 Objective

Small-scale WWTPs usually have limited resources for daily operations and especially for process upgrades. Primarily, the problems appear when available treatment capacity is not enough, or the used treatment method cannot provide proper results, and the need of operation upgrade appears.

Modelling is a tool that helps to understand the process and reach high-level optimization, which, in turn, allows to find a balance between the needs, quality and resources. However, high cost, complexity and uncertainty about the results of modelling call into question ASP modelling applicability at small-scale WWTPs. Thus, the need appears to provide a knowledge base on optimization and development possibilities of modelling use at small-scale WWTPs.

The main objective of this study is to provide recommendations on the modelling and simulation process that can be applied in the operation and process design to increase performance, and potentially reduce investments and operation costs at a small-scale WWTP where the ASP is used as the biological treatment method. The recommendations are directly addressed to the Kirkonkylä WWTP in the Lapinjärvi municipality, Finland.

1.2 Incentives

There are few incentives why the interest in applying the ASP modelling at small-scale WWTPs appears. They can be classified into two major categories:

- **Financial incentive**. Small-scale units usually do not have sufficient financial resources for daily operations or redevelopment. Prognostication of the treatment process can significantly cut the operation costs and investments to fit the planned budget by adjusting the treatment process. Moreover, the process simulation can help with WWTP layout design and unwanted extra costs by optimizing the process even before its implementation.
- **Environmental footprint incentive**. Low contaminant content level effluent is the result of a well-calibrated process. It is essential to meet effluent quality limits to obtain a permission for operation. Meanwhile,

modelling can help to achieve optimal wastewater treatment process to correspond to effluent limits in the most effective way.

Modelling of the ASP focusing on attaining improvements in the area of process design could be a valuable tool since neglecting it can result in a disruption of operations and environmental consequences.

2 BIOLOGICAL TREATMENT

Biological treatment is followed by the mechanical treatment, as shown in Figure 1. While the task of mechanical treatment is to remove large debris and fine solids from influent to ensure free movement of the wastewater through the pipes, the aim of the biological treatment can be summarised as the reduction of the total biodegradable material. The biological treatment achieves a nutrient reduction of 30–50 per cent of the total, which makes it the most substantial stage in the wastewater treatment process. (Evans & Furlong 2003, 114–115.)

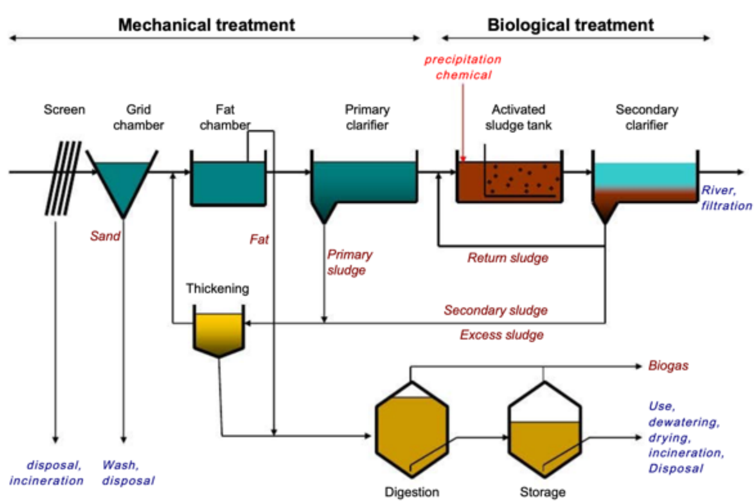


Figure 1. WWTP flowsheet (Krebs 2005)

The fundamental principle of biological treatment is the cultivation of the microorganism population in the optimised conditions leading to the reduction of the organic matter, nitrogen, and phosphorus levels. Oxidation is the fundamental basis of the microorganism growth, which is achieved in one of three types of systems, i.e. trickling filter, Wastewater Stabilisation Pond (WSP) or Activated

Sludge Reactor (ASR). ASR is considered as the primary method because of its efficiency and low cost, which is important at small-scale WWTPs. (Evans & Furlong 2003, 115–117.)

2.1 The Activated Sludge Process

In the ASP, biological treatment is achieved by the action of aerobic microorganisms. They form a functional community held in suspension, called an activated sludge, within the effluent. The aeration system provides an enhanced supply of oxygen for intensive growth. The ASP is very efficient when proper conditions for the microorganism growth are present. The treatment rate of 85 to 95 per cent for Biochemical Oxygen Demand (BOD) and Total Suspended Solids (TSS) is achieved in the ASP. (Evans & Furlong 2003, 129–132.)

The ASP provides organic matter, nitrogen and phosphorus treatment depending on the design. Only the organic carbon and nitrogen treatment are considered in the study. The excessive presence of nitrogen in effluent should be mitigated because it can lead to numerous aquatic problems such as eutrophication, aquatic organisms toxification, and the contamination of groundwater that also has a long-term effect on human health (Sutton 2011, 384–392).

The basic layout of the ASP consists of ASR and Secondly Settling Tank (SETTLER). Different configurations of ASR exist, e.g. oxidation ditch, Sequencing Batch Reactor (SBR) and Membrane Bioreactor (MBR). Every configuration differs from each other by operational parameters, e.g. Solid Retention Time (SRT), Hydraulic Retention Time (HRT), Food-to-Microorganism Ratio (F/M), technical details, and the scope of application. Nevertheless, the fundamental process mechanism is the same.

The conventional ASR configuration (Figure 2) is used to explain the basic principle of the ASP. The process begins when influent flows into ASR where a suspension of active biomass degrades organic matter with the consumption of Dissolved Oxygen (DO) and bonded oxygen atoms. Wastewater enriched with mixed liquor, composed of raw sewage and activated sludge, flows into SETTLER where mixed liquor settles to the bottom. Part of an activated sludge is recirculated back to a bioreactor as Return Activated Sludge (RAS) to maintain microorganism population and excess is removed from the circulation as Waste Activated Sludge (WAS). (Evans & Furlong 2003, 129–132.)

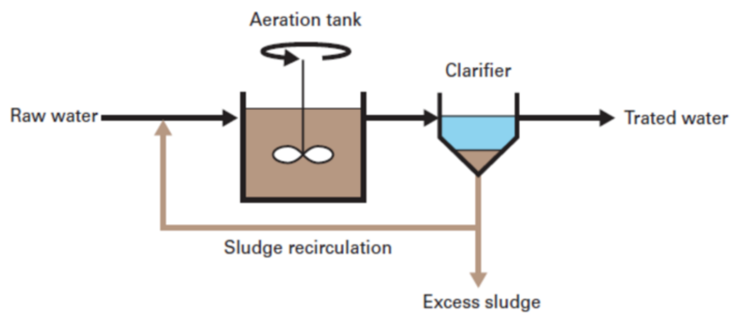


Figure 2. Configuration of the conventional ASP (SUEZ n.d.)

Activated sludge or Mixed Liquor Suspended Solids (MLSS) is a mixture of organics called Mixed Volatile Liquor Suspended Solids (MLVSS) consisting mainly of bacteria found in water, but with higher population density and inorganic particles. Bacteria need energy for growth and metabolism which they get utilizing organic matter with the consumption of oxygen. Two main types of bacteria have a role as an organic substance absorber, i.e. heterotrophs and autotrophs. The autotrophs use inorganic carbon as substrate absorbing DO. The heterotrophs use organic carbon in the form of small organic molecules as a substrate for growth, absorbing DO and bonded oxygen. (Evans & Furlong 2003, 129–132.)

2.2 Organic matter removal

Organic matter resides in the form of biodegradable and non-biodegradable matter and active biomass of microorganism in wastewater. Chemical Oxygen Demand (COD) is a measure of all oxidizable compounds, i.e. biodegradable and non-biodegradable organics, while BOD identifies the concentration of biodegradable COD only. The list of organic matter found in wastewater and its biodegradability is shown in Figure 3. (Vallero 2010.)

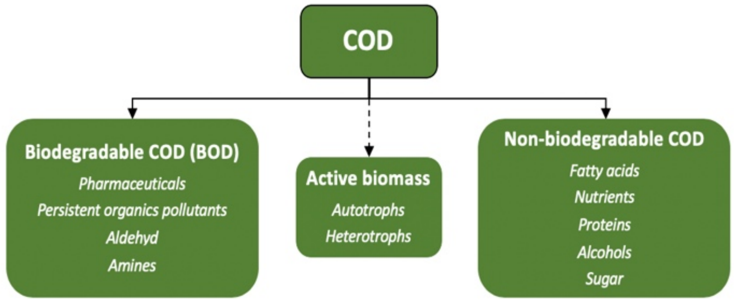


Figure 3. COD types found in wastewater (Jeppsson 1996)

Biodegradable COD is utilized as a food source for microorganisms. In contrast, non-biodegradable COD is not consumed by microorganisms and not affected by any process in the ASP. A large fraction of non-biodegradable COD settles out with the sludge, and the rest is discharged with the effluent. (Vallero, 2010.)

2.3 Nitrogen removal

Wastewater holds nitrogen in several forms, i.e. nitrite (NO_2), nitrate (NO_3) and ammonia (NH_4), and organic nitrogen, which present the Total Nitrogen (TN) content. Most of the nitrogen compounds in raw domestic wastewater are present in the form of NH_4 and organic nitrogen, and together they are called Total Kjeldahl Nitrogen (TKN), with a concentration of 60 and 40 per cent, respectively. Nitrogen is removed in two major steps, i.e. nitrification and denitrification. The nitrogen cycle is introduced in Figure 4. (Curtin et al. 2011, 13–14.)

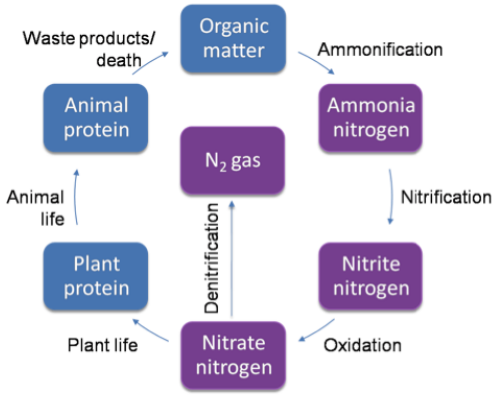


Figure 4. The nitrogen cycle, wastewater cycle is marked with purple boxes (Curtin et al. 2011)

Most of the organic nitrogen is changed to NH_4 in the assimilation process. In the nitrification and sub-step of nitrification (oxidation) processes, autotrophic bacteria, principally Nitrosomonas and Nitrobacter, derive NO_3 from NH_4. The high amount of DO is consumed for the bacteria growth, and the aerobic conditions characterized by the presence of free oxygen are necessary. The nitrification and oxidation reactions are presented as:

$$2NH_4 + 3O_2 + Nitrosomonas \rightarrow 2NO_2^- + 4H^+ + 2H_2O\ (Nitrification) \quad (1)$$

$$2NO_2^- + O_2 + Nitrobacter \rightarrow 2NO_3^-\ (Oxidation) \quad (2)$$

(Curtin et al. 2011, 13–14.)

In the denitrification process, nitrate is reduced to nitrogen gas (N_2) with release to the atmosphere as a result of heterotrophic bacteria Pseudomonas activity in an anoxic environment where the ion acts as an alternative electron acceptor to oxygen in the respiration of bacteria. The anoxic conditions characterized by the presence of atomic oxygen bounds in compounds, for example, NO_3, are necessary. The complete denitrification reaction is expressed as:

$$4NO_3^- + 4H^+ + Pseudomonas \rightarrow 2N_2 + 2H_2O + 5O_2\ (Denitrification) \quad (3)$$

(Curtin et al. 2011, 13–14.)

3 MODELLING PRINCIPLES OF ASP

In this section, the modelling principles of the ASP are explained. The modelling of ASR and SETTLER are considered separately to illustrate the process in more details. Moreover, it is explained how ASR and SETLLER are integrated into one process model.

3.1 ASR model

A set of process kinetics and stoichiometry is used to describe the biological part of the ASP. The Activated Sludge Model (ASM) family proposed by the International Water Association (IWA) incorporates process kinetics and stoichiometry into the reference biokinetic models. (Henze et al. 2000.)

There is number of biokinetic models included in the ASM family:

- **ASM1**. The Activated Sludge Model No. 1 developed by IWA Task Group (Grady et al. 1986) is the basis and framework for most models. The model illustrates the sludge production, removal of organic and nitrogenous compounds with the consumption of oxygen and nitrate ion, and it is based on 13 state variables with 8 reactions. It is originally used to estimate the required aeration capacity, the potential for denitrification, and to predict solids production for the sizing sludge handling equipment;
- **ASM2**. The Activated Sludge Model No. 2 developed by IWA Task Group (Henze et al. 1995) extends the ASM1 covering biological phosphorus treatment, and is based on 19 state variables with 19 reactions;
- **ASM2d**. The Activated Sludge Model No. 2d developed by IWA Task Group (Henze et al. 1999) extends ASM2 allowing the description of the dynamics of nitrate and phosphate, and is based on 19 state variables with 21 reactions;
- **ASM3**. The Activated Sludge Model No. 3 developed by IWA Task Group (Gujer et al. 1999 later updated in Henze et al. 2001) extends the ASM1 with the important variable as storage polymers in the heterotrophic

activated sludge conversion, and is based on 13 state variables with 12 reactions.

Numerous mathematical models of the ASP, apart from the ASM family, are reported in the literature, e.g. Barker & Dold model (Barker & Dold 1997) and UCTPHO+ (Hu et al. 2007). Nevertheless, the ASM1 is a model of choice in this study because it explains the underlying phenomena of organic matter and nitrogen treatment. Moreover, the ASM1 is still extensively used and remains the standard model, which has resulted in a long list of application cases and reported experience.

The concept of the ASM1 is explained, and the complete process kinetics and stoichiometry (the Gujer-Petersen matrix) consisting of 13 state variables and 8 process reactions is reported in Appendix 1.

3.1.1 State variables

There is a list of fundamental state variables that are used in the ASM1 for a description of wastewater content. State variables with notation and units are introduced in Table 1. Moreover, variable ranges for organic matter and nitrogen fractionation in domestic wastewater based on mean values in the EU countries, i.e. Denmark, Switzerland and Hungary, are provided (Henze et al. 2000, 25).

Table 1. ASM1 state variables (Henze et al. 1987)

No.	**Variable**	**Notation**	**Unit**	**Value range***
1	Soluble inert COD	S_I	$gCOD/m^3$	25–40
2	Readily biodegradable COD	S_S	$gCOD/m^3$	70–125
3	Inert suspended COD	X_I	$gCOD/m^3$	25–100
4	Slowly biodegradable COD	X_S	$gCOD/m^3$	100–250
5	Active heterotrophic biomass	X_{BH}	$gCOD/m^3$	n/d
6	Active autotrophic biomass	X_{BA}	$gCOD/m^3$	n/d
7	Particulate product from biomass decay	X_P	$gCOD/m^3$	n/d
8	Dissolved Oxygen	S_O	gO_2/m^3	n/d
9	Soluble nitrite and nitrate nitrogen	S_{NO}	gN/m^3	0.5–1
10	Soluble ‘ammonia’ nitrogen	S_{NH}	gN/m^3	10–30
11	Soluble biodegradable organic nitrogen	S_{ND}	gN/m^3	5–10
12	Slowly biodegradable organic nitrogen	X_{ND}	gN/m^3	10–15
13	Alkalinity	S_{ALK}	$Molar\ units$	n/d

Organic matter is divided into readily biodegradable COD (S_S), slowly biodegradable COD (X_S), soluble non-biodegradable COD (S_I), particulate non-biodegradable COD (X_I), heterotrophic biomass (X_{BH}), autotrophic biomass (X_{BA}), and biomass product (X_P) as an extra variable taking into account the cell produced from cell decay. S_S is formed of soluble molecules easily absorbed by organisms and metabolised for energy synthesis. X_S is made of complex molecules that carry out the enzymatic breakdown for absorption. In contrast, S_I is inert and not change the form through the system. X_I leaves the system at the same concentration level, particulate non-biodegradable matter turns suspended in the activated sludge, leaving with the wastewater flow. (Henze et al. 1987.)

The nitrogenous matter is divided into biodegradable and non-biodegradable. The biodegradable nitrogen is divided into soluble ammonia (S_{NH}), soluble organic nitrogen (S_{ND}), and particulate organic nitrogen (X_{ND}). X_{ND} is hydrolysed to S_{ND} in the same process with X_S. Then S_{ND} is realised as heterotrophic material and turn into S_{NH}. The autotrophic conversion of S_{NH} to nitrate is simplified to a one-step process with the consumption of oxygen. Thus, the need in addition to another characteristic arises to present nitrate and nitrogen compounds (S_{NO}). The non-biodegradable part of the nitrogenous matter is associated with the non-biodegradable COD and follows the same path with one difference: the soluble part is not incorporated into the model as it is negligible. DO consumption (S_O) and alkalinity (S_{ALK}) characteristics are included in the ASM1 as well. The inclusion of S_{ALK} is preferable as the conversion process is not compulsory, but it provides information about pH changes. (Henze et al. 1987.)

State variables act upon the process and are essential in the ASM1, but they are not always practically measurable. Characteristics measured in reality, e.g. COD, TN, TSS, are used to combine them, as shown below:

$$COD = S_I + S_S + X_I + X_S + X_{BH} + X_{BA} + X_P \ [gCOD/m^3]$$
$$TN = S_{NO} + S_{NH} + S_{ND} + X_{ND} + i_{XB}(X_{BH} + X_{BA}) + \ i_{XP}(X_P + X_I) \ [gN/m^3] \qquad (4)$$
$$TSS = 0.75(X_S + X_P + X_I) + 0.9(X_{BH} + X_{BA}) \ [gSS/m^3]$$

(Jeppsson 1996.)

The equations with the coefficients for the inert and particular matter 0.75 $gSS/gCOD$, and active heterotrophic and autotrophic biomass 0.9 $gSS/gCOD$, and parameters i_{XB}, i_{XP} $[gN/gCOD]$ based on many different municipal wastewaters are reported by Henze et al. (1987) and Jeppsson (1996), and explained in Appendix 1.

3.1.2 Parameters

Besides state variables, kinetic and stoichiometric parameters have an essential role in the ASM1 computations. The selection of parameters is known as the model calibration. Overall, 19 parameters are included in the ASM1. Parameters are wastewater specific, but some of them show little variation and may be considered to be constants.

The collection of data from WWTPs is a challenging procedure, and it is not always possible to determine the needed parameters for calibration. Thus, values may be assumed rather than evaluated for each situation. It should be stated that the parameters strongly depend on the environmental conditions. A number of environmental factors influence values and two deserve specific monitoring temperature and pH. The parameters and their standard values were introduced by Henze et al. (1987) to simplify its selection. A full list of kinetic and stoichiometric parameters and their default values are introduced in Appendix 2.

3.1.3 ASM1 Processes

Processes occurring in the ASP are incorporated into the ASM1. Processes involving bacteria, i.e. heterotrophs and autotrophs, and reaction mechanisms are shown in Table 2.

Table 2. ASM1 processes (Henze et al. 1987)

No.	Process	Reaction
1	Aerobic growth of heterotrophs	$S_S + S_O + S_{NH} \rightarrow X_{BH}$
2	Anoxic growth of heterotrophs	$S_S + S_{NO} + S_{NH} \rightarrow X_{BH}$
3	Aerobic growth of autotrophs	$S_O + S_{NH} \rightarrow X_{BA} + S_{NO}$
4	Decay of heterotrophs	$X_{BH} \rightarrow X_P + X_S + X_{ND}$
5	Decay of autotrophs	$X_{BA} \rightarrow X_P + X_S + X_{ND}$
6	Ammonification of soluble organic nitrogen	$S_{ND} \rightarrow S_{NH}$
7	Hydrolysis of entrapped organics	$X_S \rightarrow S_S$
8	Hydrolysis of entrapped organic nitrogen	$X_{ND} \rightarrow S_{ND}$

The description of the processes considered in the ASM1 is provided below:

- **The aerobic growth of heterotrophs**, the dominant share of new biomass production and removal of COD occur. X_{BH} is produced in the result of S_S utilization with intensive S_O consumption. S_{NH} incorporated into the cell mass is used as a source of energy in synthesis. Monod kinetics is used to model the growth rate.
- **The anoxic growth of heterotrophs**, X_{BH} is produced in the result of S_S utilization in the absence of S_O with S_{NH} consumption as a source of energy for synthesis. Monod kinetics is used, but anoxic conditions contribute to a lower maximum rate of S_S and correction factor η_g (0.6–1.0) is included into account.

- **The aerobic growth of autotrophs,** X_{BA} and S_{NO} are formed with the consumption of S_O and S_{NH} as an energy source. Monod kinetics is applied in the process.
- **The decay of heterotrophs**, X_{BH} formed in the aerobic growth of heterotrophs and anoxic growth of heterotrophs decompose to X_P, X_S, and X_{ND} as a result of microorganism death. Death-regeneration approach (Dold et al. 1980) is used for process modelling.
- **The decay of autotrophs**, the same mechanism and modelling approach with the same products are involved as the decay of heterotrophs.
- **The ammonification of soluble organic nitrogen**, S_{ND} is converted into S_{NH} by heterotrophs action.
- **The hydrolysis of entrapped organics**, sludge mass containing X_S is broken down to S_S under aerobic and anoxic (the correction factor η_g is used) conditions. Reaction kinetics are used for process modelling.
- **The hydrolysis of entrapped organic nitrogen**, X_{ND} is turned into S_{ND} at the same rate as in hydrolysis of entrapped organic.

The decomposition of COD and nitrate happens under aerobic and anoxic conditions which leads to different process rates since processes depend upon an electron acceptor type. The switching function concept is used in the ASM1 to adjust process rate equations when conditions vary. The oxygen half-saturation coefficient (hsc) for heterotrophs (K_{OH}) and oxygen hsc for autotrophs (K_{OA}) represent the rate of reactions. Processes which occur when S_O is present follow the following switching functions:

$$\frac{S_O}{K_{OH}+S_O}\ (heterotrophs)$$

$$\frac{S_O}{K_{OA}+S_O}\ (autotrophs) \qquad (5)$$

(Henze et al. 1987.)

In opposite, only heterotrophs participate in anoxic processes. When processes with the absence of DO modelled, the following switching function is adopted:

$$\frac{K_{OH}}{K_{OH} + S_O} \; (heterotrophs) \qquad (6)$$

(Henze et al. 1987.)

3.2 SETTLER model

SETTLER has an essential role in the ASP, and it has concurrent two functions of clarification and thickening. In more complicated cases, its functionality is extended to sludge storage and reactions place, for example, denitrification. The one-dimensional model approach is widely applied in conjunction with the ASM1, and it is used in the study. The diagram representing the one-dimensional SETTLER model is shown in Figure 5. (Makinia 2010, 126–130.)

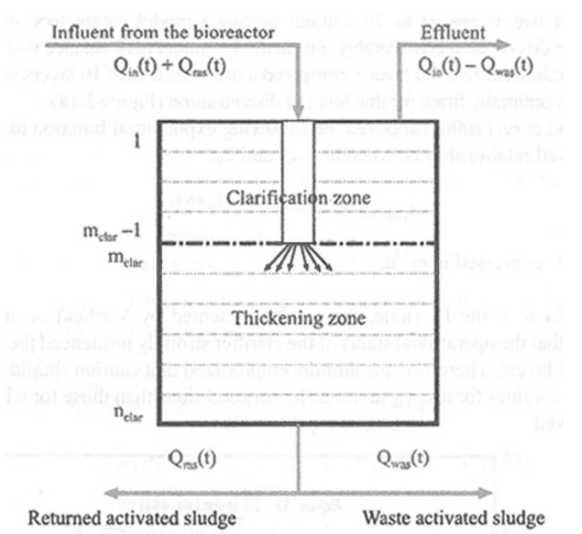


Figure 5. Diagram of 1-D model of the secondary settling tank (Makinia 2010)

The approach considers the vertical dimension modelling of the settling tank chamber while horizontal velocities and gradients are assumed to be uniform and negligible. All the soluble state variables and only one particulate state variable (solids concentration) are considered in the model, which means no settling is considered. (Makinia 2010, 126–130.)

Solids concentration as a part of inflow stream is instantaneously and uniformly spread across the horizontal cross-sectional area of SETTLER. The flow is divided into upward flow towards the effluent exit at the top and downward flow towards the underflow outlet at the bottom. Numerous layers responsible for different functions imply dividing the vertical dimension into a number of layers with the solid balance. Generally, there are five groups of layers depending on the position relative to the feed point located in the middle of the horizontal cross-section: top layer, the layers above the feed point, the feed layer, the layers below feed point and the bottom layer which can be divided to numerous sublayers. (Vitasovic 1986.)

The number of layers is an important parameter in the layered SETTLER modelling. For example, Jeppsson and Diehl (1996) proposed split SETTLER horizontal cross-section into 30–50 layers when diffusion is considered, which significantly increases the complexity of computation. In contrast, Takács et al. (1991) model with 10 layers is used in the study for the overall representation of SETTLER modelling process. It implies the calculation of mass balance, flows direction as a part of solid fluxes for each layer.

Solids entered to SETTLER move to the bottom layer and reverse under the influence of forces explained in the solid flux concept, i.e. the gravity settling flux (J_s) resulting from sludge settling and the bulk flux (J_b) resulting from moving water by a sludge recycle pump. The gravity settling flux and the bulk flux can be expressed as the product of the solid concentration (X) and the settling velocity (v_s) of the solids or bulk velocity (v_b) of the liquid, respectively. The sum of J_s and J_b gives the total flux (J):

$$J = J_s + J_b = v_s X + v_b X \qquad (7)$$

(Takács 1991.)

The solids flux applies to both flocculant sedimentation and hindered sedimentation conditions, and it is necessary to take into account the solid fluxes as well as mass balances and general flow direction. The solids fluxes with mass balances and general flows between major group layers are illustrated in Figure 6.

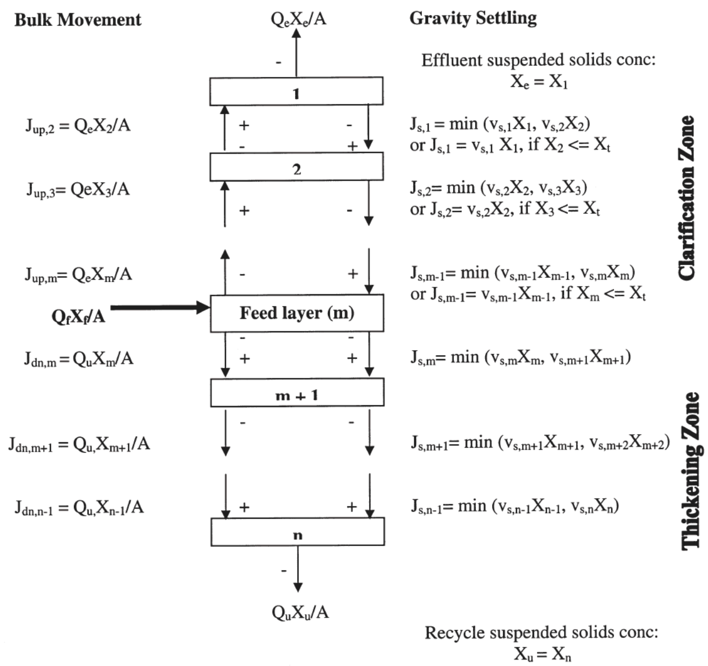


Figure 6. One-dimensional layer model by Vitasovic (1985) with equidistant layers and a constant cross-sectional area (Takács et al. 1991)

Considering J_s, the determination of v_s is necessary for proper process modelling. Numerous model proposals have been released relating v_s to X correlation, for example, by a power function or an exponential function of X. Vesilind (1968) proposed expression $v_s = v_0 e^{-cX}$ where the maximum settling velocity (v_0) and a specific model parameter (c) are calibrated individually for each layer which is labour-intensive. Later, Takás et al. (1991) proposed to express v_s as a double exponential velocity. v_s is computed for every layer (j):

$$v_{sj} = v_0 e^{-r_h(X_j - X_{min})} - v_0 e^{-r_p(X_j - X_{min})} \; [m/d]$$
$$0 \leq v_{sj} \leq v_0' \qquad (8)$$

(Takás et al. 1991.)

Where v_0 is the maximum theoretical settling velocity, $[m/d]$; v_0' is the maximum practical settling velocity, $[m/d]$; r_h is the settling parameter characteristic of the hindered settling zone, $[m^3/d]$; r_p is the settling parameter characteristic of low solids concentrations, $[m^3/d]$; $X_{min} = f_{ns}X_{in}$ is the minimum attainable suspended solids concentration, $[gSS/m^3]$, with X_{in} – the mixed liquor solids entering the settler and f_{ns} is the non-settleable fraction of X_{in}.

Thickening and clarification processes are taken into account in expressions, i.e. the large flocculating settling particles velocity is expressed in the term $v_0 e^{-r_h(x_j - X_{min})}$, and the smaller settling particles velocity is expressed in the term $v_0 e^{-r_p(x_j - X_{min})}$. The determination of v_s consequently leads to mass balances around layers. (Takás et al. 1991.)

Considering J_b, it should be taken into account that v_b may be either downward (v_{dn}) or upward (v_{up}) depending on a position relative to the feed layer:

$$v_{dn} = \frac{Q_u}{A} = \frac{Q_r + Q_w}{A}$$
$$v_{up} = \frac{Q_e}{A} \qquad (9)$$

(Takás et al. 1991.)

Where A is the settler cross-sectional area, $[m^3]$; Q_u is the underflow rate (with Q_r as recycled flowrate and Q_w as wasted flowrate), $[m^3/d]$; Q_e is the effluent flow rate, $[m^3/d]$.

3.3 ASR and SETTLER models merge

Modelling issue, for example, dynamics and state variables conversion, occur when merging ASR and SETTLER into a uniform model. The issue of coupling ASR and SETTLER dynamics occurs because of the recycle flow. The ASM1 is based on 13 state variables while only the total concentrations of the particulate and soluble material are used in the SETTLER model. Moreover, the conversion is complicated due to the different units used in the ASM1 and the SETTLER model. In the SETTLER model, the concentrations are expressed in $[gSS/m^3]$, but in the ASM1, COD is expressed in $[gCOD/m^3]$ and TN is expressed in $[gN/m^3]$. (Diehl & Jeppsson 1998.)

A set of factors has to be taken into account for the correct final outputs when converting state variables. Relevant mass for all state variables has to be computed as the particulate material tends to settle due to the gravitational force. It is assumed that DO is completely consumed within SETTLER. Soluble state variables follow the streams without settling. Moreover, it is assumed that there are no biological reactions in SETTLER, which means the conversion coefficients are not needed when converting COD into X. X_{ND} is not accounted when converting TN into X because it is included as a subset of other particulate state variables. (Diehl & Jeppsson 1998.)

4 SIMULATION OF ASP

By using simulation, it can be predicted how the ASP undergoes in the short term as well as long-term perspective. Simulation makes it possible to determine the system sensitivity to different conditions and test variety control configuration. As a result, the needed optimisation can be implemented to improve treatment outputs. Simulation can be organised in a working environment, either using ready-to-use simulation software or programming environment. Only the simulation software approach is used in the study.

4.1 Simulation software

Simulation software ties together biological, chemical, and physical process models, and allows to compare design alternatives and find the desired option in respect to an effluent quality, operational costs and investments. It can be used for daily plant operations, control strategies, plant design and even re-design. It provides extensive libraries of pre-defined and custom process models offering the representation of the whole WWTP processes. The built-in libraries allow to easily simulate the ASP and modify the model parameters using the user interface.

There is a list of widely used simulation software in the industry:

- **GPS-X™** is the first released dynamic water treatment plant simulator developed by Hydromantis Environmental Software Solutions Inc. (Canada). The libraries include native Mantis model extending the ASM1 model, which widely used for plant operation. Up-to-date information can be found on https://www.hydromantis.com/GPSX.html.
- **BioWin** is developed and supported exclusively by EnviroSim Associates Ltd. (Canada) and recognised as the original “whole-plant” model. It has a built-in ASDM model that extends IWA model libraries and used by private companies. More detailed information about the simulator can be found on https://envirosim.com/.
- **SIMBA#water** is a version of Simulink for wastewater treatment applications owned and supported by inCTRL Solutions (Canada). Besides, it provides tools for biogas plants in extra package libraries called *SIMBA#biogas*. The current progress and information about the software can be found on https://www.inctrl.com/software/simba/simbawater/.

WEST

The WEST (Wastewater treatment plant Engines for Simulation and Training) has been chosen as a simulator for the study. The WEST is a virtual simulation platform for dynamic modelling and simulation of WWTPs and other types of water quality-related systems. It was developed at the University of Gent (Belgium). To date, it is one of MIKE Powered by DHI software products which is widely used in urban modelling. The WEST is designed for plant operators, engineers, consultants and researchers, and available in four different license types, i.e. WESTforOPERATORS, WESTforDESIGN, WESTforOPTIMIZATION, and WESTforAUTOMATION, based on customer needs. In the study, WESTforOPTIMIZATION license providing all the possible ranges of functionality is used. More detailed information about the licences, versions and current development of the WEST software can be found on https://www.mikepoweredbydhi.com/products/west. (ISSUU, 2020.)

The WEST provides an extensive model library, including physical and biochemical models. With regard to ASR, the whole ASM family is integrated into the WEST library. With regard to SETTLER, the zero and the monodimensional models, including the model of Takás et al. (1991) for the reactive and the non-reactive cases, are integrated into the WEST library. Moreover, the WEST has a range of functional modules for the convenience modelling process. The WEST provides tools based on a graphical interface for a direct application to WWTPs and a possibility to reuse previously developed models.

4.2 Simulation of small-scale WWTP

Small-scale WWTPs do not generally provide many opportunities for the control of influent, nor for the control of operating parameters based on on-line measurements. Moreover, high fluctuations in the hydraulic and organic load can significantly affect the accuracy of the model simulation, especially in small systems. Nevertheless, the modelling of small-scale wastewater treatment process, with the focus on attaining improvements in the area of process design, could be a valuable tool. (Philips et al. 2000.)

The general application of modelling at small-scale WWTPs is the same as at large-scale WWTPs, but with limitations and a local use due to a lack of available data about plant operations. According to international IWA survey (2009), model building use is associated with the following objectives: optimisation (59 per cent of users), design (42 per cent of users), and prediction of future operations (21 per cent of users). (Hauduc et al. 2009.)

Small-scale WWTPs usually provide scarce data on the routine operations, which results in the limited modelling application. Nevertheless, if a reliable plant model is obtained, a list of simulation scenarios can be tested. For example, the plant operation can be assessed for the sludge production, plant capacity and nitrogen treatment. Moreover, site-specific models can be created for operation training and increasing the qualification of personnel. (Rieger et al. 2013, 97–99.)

Designing new WWTPs is a continuous and complicated process. However, small projects are usually well defined and have limited objectives. Thus, there are comparatively not many additional data collection procedures needed to establish the process design at a small-scale WWTP compared to large-scale WWTPs where many additional costly laboratory tests and efforts are needed. Thus, the ASP modelling for the process design purposes can be a valuable tool for the development of a smooth wastewater treatment process with minimum environmental impact and operation costs.

4.3 Simulation modes

Two different types of data; average and time-series, describing hydraulic and organic loads are used for steady-state and dynamic simulations, respectively. The choice between the steady-state and dynamic simulation depends on the available data and study target. Small-scale WWTPs' monitoring procedures usually provide limited data including partial data on daily, monthly or even yearly average periods. Moreover, the absence of on-line measurement of parameters limits the choice of simulation modes to steady-state.

Nevertheless, steady-state simulations are efficient at identifying data samplings or analytical errors as the mass balance is always maintained in the simulation software. Simulation results enable to identify errors from laboratory or monitoring data. Moreover, steady-state simulations are valuable to estimate WWTP performance under various scenarios. For example, operating or the future WWTP layout can be tested under different load conditions, e.g. current, future, or custom. In the process, the problems in the operations of WWTP can be identified. In opposite, steady-state simulations are not applicable for sizing equipment operating under dynamic conditions, daily behaviour, or storm handling dependent on the peak and minimum flows or loads where dynamic data is needed. (Rieger et al. 2013.)

5 MODELLING STEPS

The different modelling protocols providing step by step modelling process guidelines have been developed. The most notable protocols are BIOMATH (Vanrolleghem et al. 2003), STOWA (Hulsbeek et al. 2002; Roeleveld & Van Loosdrecht 2002), and WERF (Melcer et al. 2003). However, they are focused on different backgrounds, e.g., academic and consultancy, and different objectives. As a result, there are often insufficient instructions for a proper assessment when modellers do not choose an appropriate protocol.

5.1 The GMP Unified Protocol

The Good Modelling Practice (GMP) Unified Protocol includes the main elements from numerous analysed protocols which according to The GMP Task Group (the IWA World Water Congress 2004) should be included in the general activated sludge protocol.

The protocol comprises five major steps:

- **Step1. Project definition**, the objective is formed based on the client request. Expectations, responsibilities and the budged target are agreed.
- **Step 2. Data collection and reconciliation**, existing and missing data, i.e. input data, physical data, operational settings and performance data are collected in the process of the measuring campaign. The current hydraulic and organic load and process scheme should be compared with the original design. Reconciliation is especially important as it plays an essential role in the planning phase of the model. Typically, this step consumes about one-third of the overall time of a model-based study (Hauduc et al. in press).
- **Step 3. Plant model set up**, the simulation platform and sub-models are determined. WWTP layout is set up and the parameters for initial run are selected. The initial model should be implemented, and a number of tests should be done.

- **Step 4. Calibration and validation**, the agreement between observed and simulated data is achieved based on the stop criteria set in Step 1. The parameters are calibrated and validated using independent data if the simulated and measured data do not match reasonably.
- **Step 5. Simulation and result interpretation**, various simulations are run in order to get different outputs and meet the objective. The final product is a standardized report allowing the client to compare the results and get a clear understanding of the future simulations and model use. (Rieger et al. 2013)

It is worth mentioning that each step is agreed upon with the client before the next step is started. The protocol step by step structure is illustrated in Figure 7.

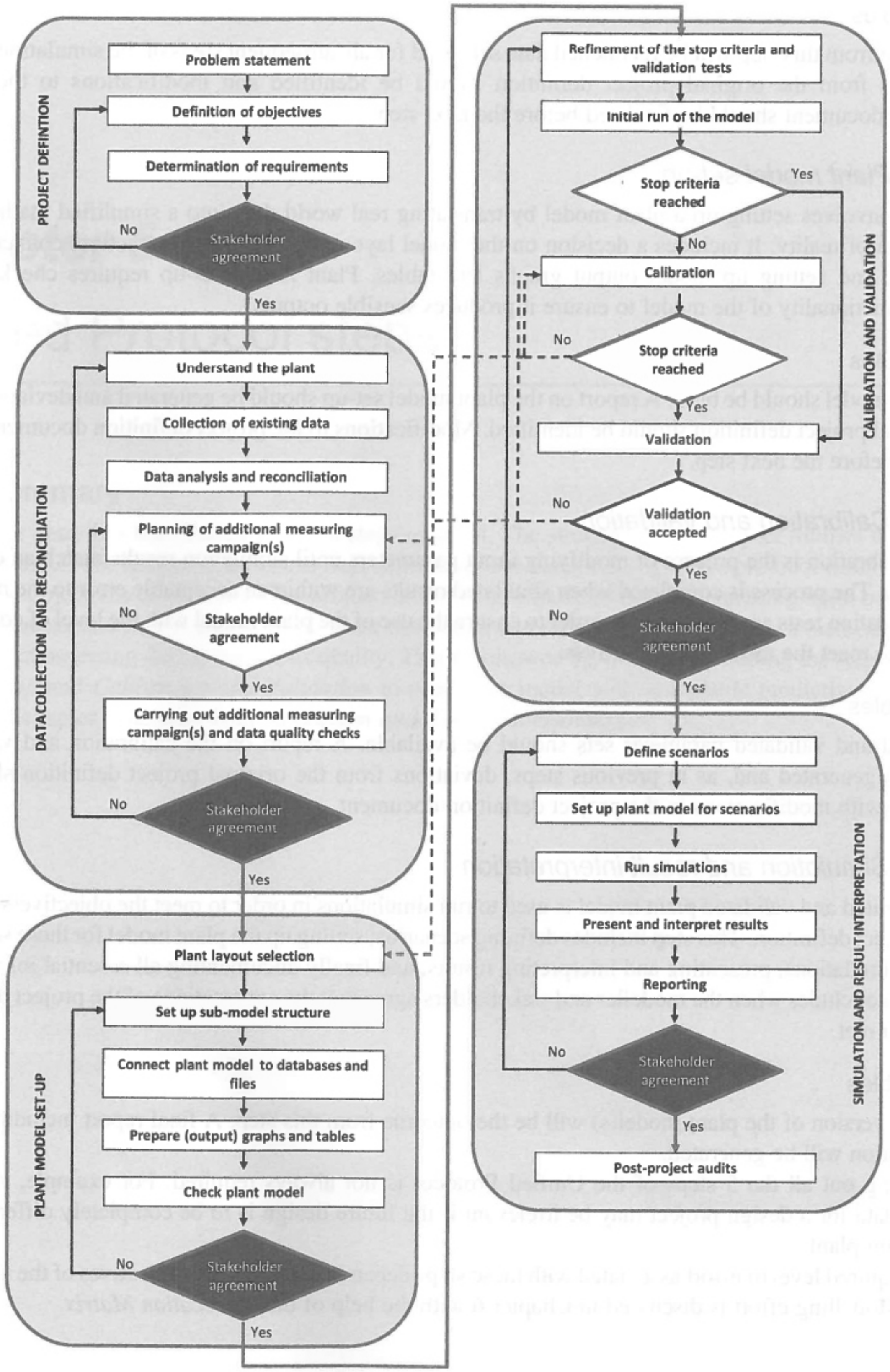


Figure 7. Schematic representation of the GMP Unified Protocol (Rieger et al. 2013)

6 CASE STUDY: WWTP IN LAPINJÄRVI

The initiation of the case study including the problem statement and the pursued objective is explained in this Chapter. The wastewater treatment situation in the Lapinjärvi municipality in considered in general. The situation at the Kirkonkylä WWTP with a scope on the existing operation issues is discussed.

6.1 Problem statement

Lapinjärvi municipality is located in the Uusimaa region, Finland, with a population of approx. 2,800 people (Väestörekisterikeskus 4/2014). To date, the municipality faces difficulties in the wastewater treatment due to the increased need in capacity and the ageing wastewater treatment facility (Luoma-Aho 2020).

There are three WWTPs providing wastewater treatment in the municipality: the Kirkonkylä, Siviilipalveluskeskus and Porlammi. The Kirkonkylä WWTP, which is the object of the case study, is located on the east of Lapinjärvi village centre (Lapinjärven Kirkonkylä) and constructed in 1976, see Figure 8. The facility treats liquid waste disposed by local households (100 per cent domestic wastewater) connected to the sewerage system and discharged to Taasianjoki river through Hölkesbäcken canal. (Luoma-Aho 2020.)

Figure 8. The Kirkonkylä WWTP location

The Kirkonkylä WWTP has been designed to process wastewater flows of 300 m^3/d. To date, households with approx. 2,000 people are connected to the

WWTP through the sewer system (Luoma-Aho 2020). Domestic households in Europe produce an average of 0.15 m^3/d of wastewater per capita (Ruokojärvi 2007). Thus, the population equivalent of 2,000 produces approx. 300 m^3/d of wastewater. Nevertheless, the hydraulic load fluctuates and reaches up to 1,800 m^3/d during winter-spring seasons because of stormwater. The yearly average hydraulic load of 389 m^3/d was reported in 2019 (Ramboll, 2019). Consequently, the difficulty with the processing of inflow water which decreases treatment efficiency during intensive precipitation season appears.

The treatment efficiency is worsened because of the deteriorated pipes, which leads to infiltration/exfiltration at the Kirkonkylä WWTP (Luoma-Aho 2020). It leads to the decrease of COD (low F/M) and high inorganic solids which result in high treatment cost, poor resource recovery, additional volume requirement of ASR, and SETTLER and other operational issues (Cao et al. 2019).

Siviilipalveluskeskus and Porlammi WWTPs operate in the municipality to cope with the wastewater disposed of by distant households from Lapinjärvi village centre. The sewer systems are not connected with each other. On average, they processed 223 m^3/d of wastewater cumulatively in 2019, which is 57 per cent of the Kirkonkylä WWTP hydraulic load.

Consequently, the need for the WWTP upgrade, due to the increased hydraulic and organic loads, obsolete equipment and operating methods, coupled with the lack of safety and environmental policy, has appeared. The proposal to construct a new unit that will replace the Kirkonkylä WWTP and possibly all WWTPs operating in Lapinjärvi municipality is highlighted (Luoma-Aho 2020).

6.2 Objective

The objective of the simulation study is an evaluation of the ASP modelling applicability at the small-scale WWTP, and study of the feasibility of setting up the model based only on the typically available operational and design data without performing additional data collection procedures. Besides, the ASP modelling process and possible benefits for process planning and improvement at the small-scale WWTP are demonstrated.

The possible application cases of the ASP modelling use are demonstrated by simulating the Kirkonkylä WWTP layout model. The simulation of mechanical treatment and sludge processing is not performed, and the boundaries are limited with the biological treatment. The inclusion of the mechanical treatment and sludge processing models into the WWTP layout model increases the complexity substantially, which, considering the limited set of data, may lead to unreliable simulation results.

The modelling procedure involves steady-state simulations to predict long-term performance and get an overall observation of the modelling use at the small-scale WWTP with the scope on organic matter and nitrogen treatment. No stop criterion for the model calibration is set as the lack of data and inaccuracy related to ageing equipment cannot be predicted as a part of this study. Nevertheless, the model calibration is performed to get the simulation results as close as possible to the measured and satisfy the margin of 5–15 per cent accuracy established in GMP Unified Protocol guidelines.

7 MATERIALS

In this Chapter, the data for simulation study based on routine operational data collected during the operation year 2019 by Ramboll and Eurofins from the Kirkonkylä WWTP are provided and analysed. Besides, the construction data from WWTP design sources, i.e. WWTP master plan and sizing description, is considered. No additional measuring campaigns were organised.

7.1 Process flow

The modified structure of the conventional ASP called oxidation ditch is used at the Kirkonkylä WWTP. The system consists of ASR (a single oval-shaped channel) showed in Figure 9a, and SETTLER (Dortmund type tank composed of four chambers) showed in Figure 9b. The treatment of organic matter and nitrogen is obtained with the ASP, where only the aerobic conditions are established to achieve nitrification and the anoxic conditions for denitrification are not achieved. Horizontally mounted brush aerators and fine-bubble diffusers are used to provide aeration, oxygen transfer and circulation in ASR.

(a)

(b)

Figure 9. The Kirkonkylä ASP; (a) Oxidation ditch, (b) Secondary settling tank

The combination of oxidation ditch and SETTLER, called the continuous flow type oxidation ditch, provides the continuous process by allowing the mixed liquor to settle in SETTLER, see Figure 10. The settled sludge is returned from the bottom of SETTLER to ditch through the RAS pipeline. The oxidation ditch is a closed system, and the excess of sludge is removed through WAS pipes connected to the RAS pipeline to sustain more active metabolism ration. (Punmia 1998, 447–450.)

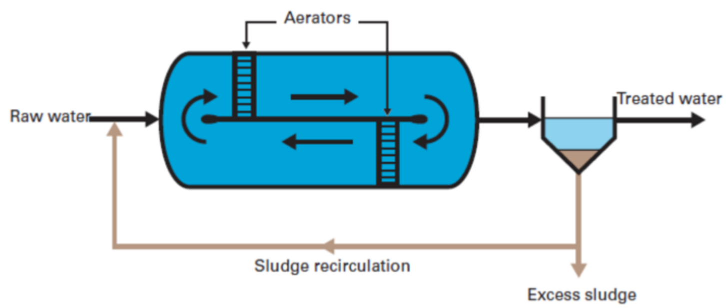


Figure 10. Configuration of continuous oxidation ditch (SUEZ n.d.)

Oxidation ditch is a complete mix system with long SRT varying from 4 to 48 days and low F/M in range of 0.05 to 0.15 $gBOD/gSS$ for nitrification and denitrification. The configuration has several advantages, e.g. efficient energy consumption and low production of activated sludge, if the process is adjusted correctly. Contrary, MLSS concentration is high, and a ditch placing requires a large land area for a ditch placing compared to other ASP modifications. (EPA 2000.)

Chemicals dosing is applied at the Kirkonkylä WWTP to enhance phosphorus removal. Ferrous sulfate is used with lime to form calcium sulfate and ferric hydroxide, which has a high phosphate adsorption capacity. (EPA 2000.) Nevertheless, chemical dosing is not considered in the modelling procedure as it is used for phosphorus removal, which is not considered in the study, and does not affect the organic matter and nitrogen treatment at the Kirkonkylä WWTP.

7.2 Input data

The hydraulic load is monitored and recorded on a daily basis as a part of a standard monitoring procedure. Monthly and yearly averages of the hydraulic load during the monitoring year 2019 are introduced in Table 3.

Table 3. Hydraulic load (Ramboll 2019)

Month	Hydraulic load			
	Minimum $[m^3/d]$	Average $[m^3/d]$	Maximum $[m^3/d]$	Total $[m^3/month]$
January	307	360	445	10,073
February	284	476	826	12,856
March	277	620	1,542	19,213
April	296	558	1,542	16,754
May	294	504	1,398	15,619
June	207	264	335	7,404
July	61	245	475	7,589
August	171	278	414	8,613
September	141	188	268	5,655
October	158	230	434	9,288
November	226	390	1,184	11,700
December	336	559	1,867	17,337
Total				142,101
Yearly average				389

The organic load is monitored as a part of the environmental permit monitoring procedure. The data is limited with one-two days sampling periods during a month, with a total of seven sampling months. The measurements for each monitoring month and yearly average values of influent organic composition during monitoring year 2019 are introduced in Table 4.

Table 4. Influent organic composition (Eurofins 2019)

Parameter	Concentration $[\mathbf{g}/\boldsymbol{m}^3]$	Minimum percentage of reduction $[\%]$
COD	125	75
BOD_7	30	70
TSS	35	90
TN	15	70

7.3 Physical data

The technological ASP layout consists of oval-shaped oxidation ditch (with the total volume of $V_{ASR} = 340\ m^3$, depth of water line $H_{ASR} = 1.2\ m$) and four Dortmund type tanks (with the total surface area $A_{SETTLER} = 48\ m^2$, depth

$H_{SETTLER} = 6\ m$) connected with RAS pipeline operating as conventional activated sludge treatment with the low release of WAS.

Two mounted surface brush aerators and fine-bubble diffusers are installed in ASR to mix wastewater and supply oxygen. Fine-bubble diffusers and one of the surface aerators are in the normal operation mode, and the second surface aerator is plugged in when more oxygen supply and mixing is needed.

According to the design data, internal flow rates are the following. Wastewater overflows to SETTLER with overflow rate (Q_o) equal to inflow rate (Q_i). RAS is returned to oxidation ditch through RAS pipeline by pumps. Four airlift pumps NS 50 with piston compressor (2.4/1.2 Nm^3/min x 0.44 bar) procced 300 m^3/d of RAS flow (Q_r) which is equal to the design inflow rate (Q_{di}). The excess sludge is released through WAS pipe connected to RAS pipeline by a screw-type pump (1 m^3/h x 1.47 bar) with WAS flow rate (Q_w) of 3 m^3/d to the sludge storage where it is aerated.

7.4 Performance data

Effluent organic composition data were obtained as a part of the environmental permit monitoring procedure. The measurements for each monitoring month and yearly average values of effluent organic composition during monitoring year 2019 are introduced in Table 5.

Table 5. Effluent organic composition (Eurofins 2019)

Parameter	**Sampling month**							
	Feb.	**Mar.**	**May**	**June**	**Sept.**	**Oct.**	**Nov.**	**Avg.**
COD $[g/m^3]$	34	27	41	27	36	38	82	40.7
BOD_7 $[g/m^3]$	4.3	2.6	5.8	4	6.5	3.2	13	5.6
TN $[g/m^3]$	37	21	41	44	60	54	34	41.6
TSS $[g/m^3]$	4.8	20	5.8	8.7	31	14	31	16.5
NH_4 $[g/m^3]$	30	17	31	1.5	18	12	19	18.4

According to the minimum requirements for biological wastewater treatment declared in Government Decree on Urban Waste Water Treatment 888/2006, BOD_7 must be ≤ 15 g/m^3, TSS ≤ 35 g/m^3, TN ≤ 15 g/m^3 for effluent with a treatment efficiency of 70, 90 and 70 per cent, respectively. Effluent BOD_7 and TSS yearly average concentrations of 5.6 g/m^3 and 16.5 g/m^3 with removal rates of 96 per cent and 91 per cent in 2019 meet the declared limits, but effluent TN yearly average concentration exceeds the limit with the value of 41.6 g/m^3 and poor removal rate of 18 per cent.

The yearly average TN load of 19.8 kg/d in 2019 exceeded the designed load of 14.4 kg/d. In opposite, BOD_7 load was low of 55.9 kg/d compared to the designed load of 84 kg/d and the yearly average value of 80 kg/d in 2018, which can be explained by deteriorated pipes, which leads to infiltration/exfiltration.

DO concentration is measured on a daily basis in ASR and SETTLER as a part of a standard monitoring procedure. The monthly and yearly averages of DO concentration during 2019 are introduced in Table 6.

Table 6. DO concentration in ASR and SETTLER (Ramboll 2019)

Month	**DO concentration $[mg/l]$**	
	ASR	SETTLER
January	8.3	8
February	7.6	7.4
March	8.6	8.4
April	7.6	7.7
May	3.9	5
June	1.9	1.9
July	3.8	1.8
August	3.2	2.6
September	3.1	2
October	5	4
November	6.5	6
December	7.7	7
Average	5.6	5.2

The presence of high DO concentration in SETLLER implies that SETTLER is non-reactive, and no denitrification occurs.

Additionally, the yearly average values for influent and effluent in 2019 of the following parameters were measured: temperature ($T_{inf} = 10.6\ ℃$; $T_{eff} = 11.7\ ℃$), pH ($pH_{inf} = 7.4$; $pH_{eff} = 6.5$), alkalinity ($Alk_{eff} = 0.72\ mmol/l$) and conductivity ($\kappa_{inf} = 126.8\ mS/m$; $\kappa_{eff} = 101\ mS/m$).

To date, the Kirkonkylä WWTP operates on its maximum designed hydraulic and organic load. Most time of the year it is overloaded, which significantly affects the effluent quality. According to the wastewater treatment plant annual report 2018, with regard to the ASR and SETTLER volumes, the Kirkonkylä WWTP can operate more efficiently at a higher load compared to the design values (Ramboll 2019).

8 RESULTS AND ANALYSIS

In this Chapter, the applicability of ASM modelling at small-scale WWTP with limited data is demonstrated. The step-by-step model building and simulation Scenarios are explained. Moreover, the recommendations on optimisation and the future development of the Kirkonkylä WWTP by using the ASP modelling are provided. The simulation study is organised adopting the the GMP Unified Protocol guidelines

8.1 Plant model set-up

A reliable WWTP layout model was set up based on the collected operation data in the WEST software environment. The ASM1Temp WEST instance was chosen as a biokinetic model due to its relatively low complexity, which was essential for calibration and validation since the collected data was limited. The hydraulic behaviour was represented by ideal Continuous Stirred Tank Reactors (CSTRs) having a uniform concentration within its confines, which means influent is mixed wholly and instantaneously.

8.1.1 Plant layout

The layout model was created using the list of available WEST blocks, i.e. municipal wastewater, activated sludge tanks, secondary clarifier, effluent, waste, PI controllers, and flow splitters and combiner, representing the graphical model of a real-life process (Figure 11).

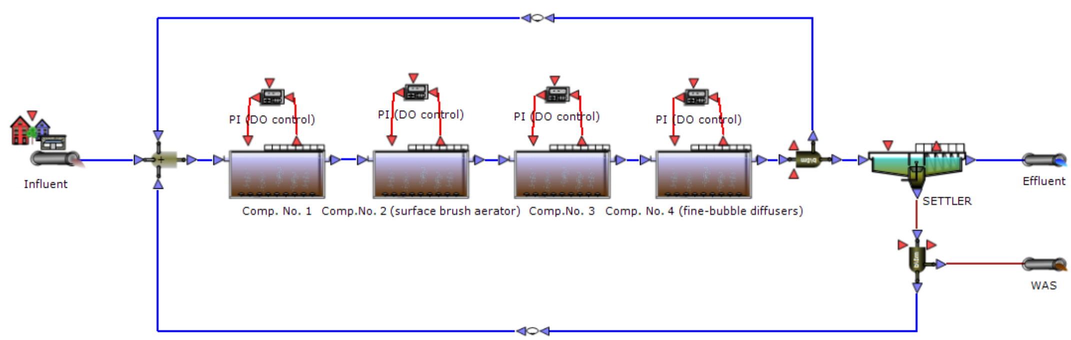


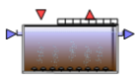
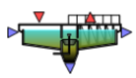
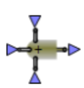
Figure 11. The Kirkonkylä WWTP model layout in the WEST software

Oxidation ditch was represented as four upstream Activated Sludge (AS) tanks alternating aerobic compartments with different DO concentration. It was assumed that DO concentration cannot be at the same level within the oxidation ditch due to mixing and oxygen transfer. The oxidation ditch was divided into four compartments: two compartments with aeration equipment where DO concentration is high DO (measured) and two compartments without aeration equipment with lower DO concentration. DO concentration in compartments was fixed at the specified levels by a Proportional-Integral (PI) controllers which are altering the value of the oxygen mass transfer coefficient ($K_L a$) to reach constant DO concentrations. The recirculation stream from AS tank Comp. No. 4 to Comp. No. 1 represents the natural wastewater circulation in the oxidation ditch with the flow rate of Q_i.

8.1.2 Sub-model structure

The WEST blocks have independent sub-models with a set of rules for dynamics and state variables conversion from sub-model to sub-model and interaction with the ASM1 parameters and state variables. The full list of used blocks and sub-models is introduced in Table 7.

Table 7. The list of sub-models (MIKE 2017)

Block type	**Sub-model**
Municipal wastewater 	The ASM1 fractionation model
Activated sludge (AS) tank 	CSTR with constant volume model
Secondary clarifier 	Layered model by Takács based on the model of Vitasovic
PI controller 	Transfer function model
Effluent 	The ASM1 defractionation model
Waste 	
Flow splitter/combiner  	Pumped flow model

The default fractionation model was modified by adding input component NO_x (equals S_{NO} state variable) to indicate nitrate concentration (see Table 1) in influent as the standard WEST ASM1 fractionation model sets S_{NO} to 0 gN/m^3.

8.1.3 Connection to databases

The collected data were examined to extract several sets of input data representing 30-day periods of steady-state operation. The data were elaborated and averaged, assuming that this average represents a steady-state operation. Two data sets were recorded in spreadsheets and used for model calibration and verification: (a) 2019 yearly average data set, (b) October average data set (the maximum organic load). The input data sets and related settings are introduced in Table 8.

Table 8. The data sets for steady-state simulation

Parameter	**Data set**			
	(a) Yearly average		(b) October	
Influent flow rate (Q_i) $[m^3/d]$	389		230	
RAS flow rate (Q_r) $[m^3/d]$	300			
WAS flow rate (Q_w) $[m^3/d]$	3			
DO concentration in compartments (from No. 1 to No. 4) $[mg/l]$	2; 5.6; 2; 5.6		2; 5; 2; 5	
Concentration of	Influent	Effluent	Influent	Effluent
COD $[g/m^3]$	387.1	40.7	430	38
TN $[g/m^3]$	51	41.6	66	54
NH_4 $[g/m^3]$	n/d	18.4	n/d	12
NO_x $[g/m^3]$	0.5	n/d	0.5	n/d
TSS $[g/m^3]$	181.4	16.5	180	14

The blocks were adjusted with the following physical parameters and operational settings: AS tanks (volume of tanks from No. 1 to No. 4: 120 m^3; 50 m^3; 120 m^3; 50 m^3), PI controllers (DO concentration according to the data set settings, see Table 8), secondary clarifier (underflow rate 300 m^3/d, surface area 48 m^2, height 6 m), flow splitter to recirculation (flow rate equal Q_i according to the data set settings, see Table 8), flow splitter to WAS (flow rate 3 m^3/d).

8.1.4 Calibration

The initial run using the data set (a) as input with automatically generated fractionation state variables (Table 9) and parameters typical for 10 ℃ and neutral pH (Appendix 2) was executed to compare the simulation values with the observed values. The simulation run was continued until stable nitrification is reached. The results of the data set (a) calibration in the WEST software are provided in Appendix 3.

Table 9. Fractionation state variables of input datasets

	Organic matter $[gCOD/m^3]$					Nitrogen $[gN/m^3]$			
	S_I	S_S	X_I	X_S	X_{BH}	S_{NO}	S_{NH}	S_{ND}	X_{ND}
Data set (a)	36.3	108.9	36.3	181.4	24.2	0.5*	33.15	7.1	10.7
Data set (b)	47.5	142.5	36	180	24	0.5*	42.9	9.2	13.9
* – set by adding NO_x component in fractionation model and mentioning in the input data spreadsheets									

Comparing the simulated and the observed data, the discrepancy in the values was identified (Figure 12 (default)). The significant difference in NH_4 content in effluent appeared. This may be related to a list of possible faults in the modelling and treatment operation sides.

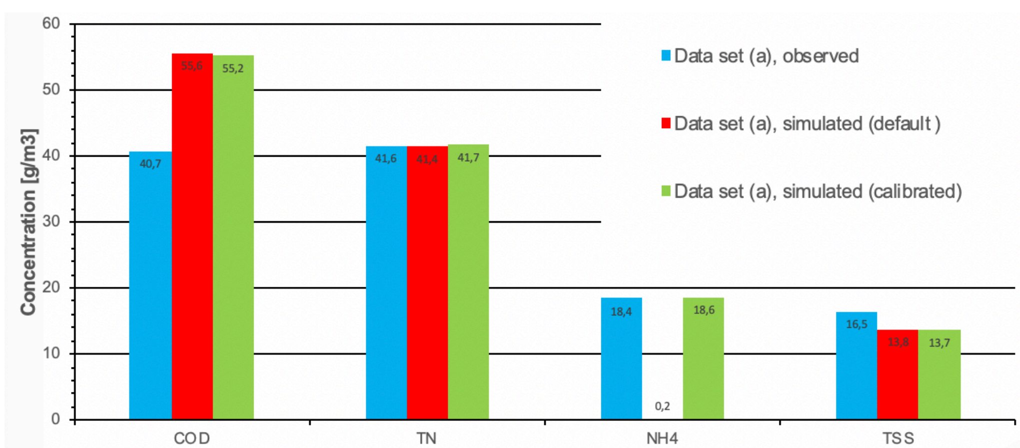


Figure 12. Results of model calibration

The inability to precisely estimate the DO concentration in every part of the oxidation ditch affects the prediction of the nitrification rate that leads to NH_4 treatment simulation inaccuracy. Identified infiltration/exfiltration reduces COD content and therefore lowers F/M, which affects TN treatment. Finally, the fluctuation in the influent composition and harsh operating conditions due to temperature differences during cold and warm seasons can significantly affect simulation prediction of small systems. Nevertheless, the calibration of the parameters should be performed to improve model prediction results.

The calibration was done in a steplike manner on the basis of the acquired knowledge of the ASP modelling. The calibration was continued until a satisfactory, from a modeller perspective, a match between simulated and observed values was reached. The instructions provided in the GMP Unified Protocol (Rieger et al. 2013) were taken into account during the calibration procedure. As a result of the calibration, the following ASM1 kinetic parameters were adjusted: K_{NH}, k_a, b_A.

The parameter K_{NH} was increased from 1 to 9 $gNH_3 - N/m^3$. K_{NH} is a switching function to stop nitrification in case of low substrate concentration. The increase of K_{NH} value implied a loss in substrate concentration due to deteriorated pipes causing infiltration/exfiltration. The parameter b_A was increased from 0.1 to 0.15 $1/d$. The adjustment of this kinetic parameter leads to decrease of the nitrification velocity and the decrease of NH_4 to NO_3 ratio in the effluent. The parameter k_a was decreased from 0.04 to 0.02 $m^3/gCOD/d$. The decrease of k_a value leads to more organic soluble substances containing nitrogen remaining in effluent because the ammonification rate is slower due to temperature fluctuations.

As a result of the calibration of the parameters, the quality of the simulation prediction was significantly improved, see Figure 12 (calibrated).

8.1.5 Validation

The data set (b) was simulated to verify the predictive accuracy of the calibrated model. The adjusted parameters were kept the same as in the calibration, but the operational settings were changed according to the data set settings (Table 8). The results of the model validation are illustrated in Figure 13. The results of the data set (b) validation in the WEST software are illustrated in Appendix 3.

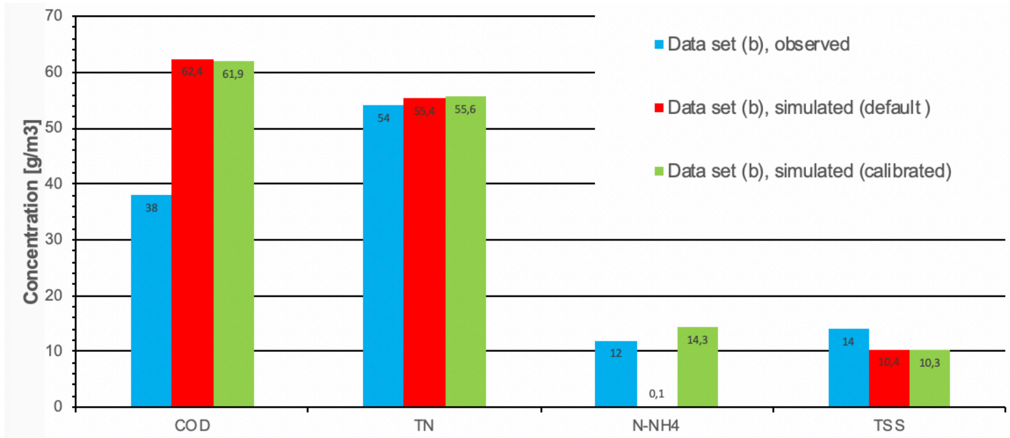


Figure 13. Results of model validation

As a result of the validation run, it can be stated that the plant model is reliable, and the simulated values for the nitrogen treatment are within an acceptable error range of 5–15 per cent (Rieger et al. 2013). However, the COD concentration difference is high, which can be related to treatment operation faults such as infiltration/exfiltration due to the ageing equipment. The ASM1 does not provide the possibility to include this aspect and extensions are needed. The objective of the study was to show the modelling application at small-scale WWTP, and detailed modelling of side aspects such as infiltration/exfiltration is not considered. Nevertheless, the model can be used to predict the general performance trend of the Kirkonkylä WWTP.

8.2 Simulation and result interpretation

The calibrated model was applied to understand and study the technological process at the Kirkonkylä WWTP. The basic operational parameters were calculated using the WEST software instruments. Moreover, two scenarios were simulated to predict the system performance in different circumstances: Scenario No. 1: Potential treatment performance; Scenario No. 2: Operation under increased hydraulic load. The data set (a) representing 2019 yearly average load was used. The results are shown and discussed in the following sections.

8.2.1 Process operation observation

The operational parameters such as SRT, HRT and F/M were calculated using the process calculator block. Besides, sensors were added in the model layout to take a reading of parameters needed for calculations, see Figure 14. The following data was fed into the process calculator block: volumes and MLSS of AS tanks, flow rate and TSS concentration of effluent and WAS, and flow rate, COD and BOD concentrations of influent.

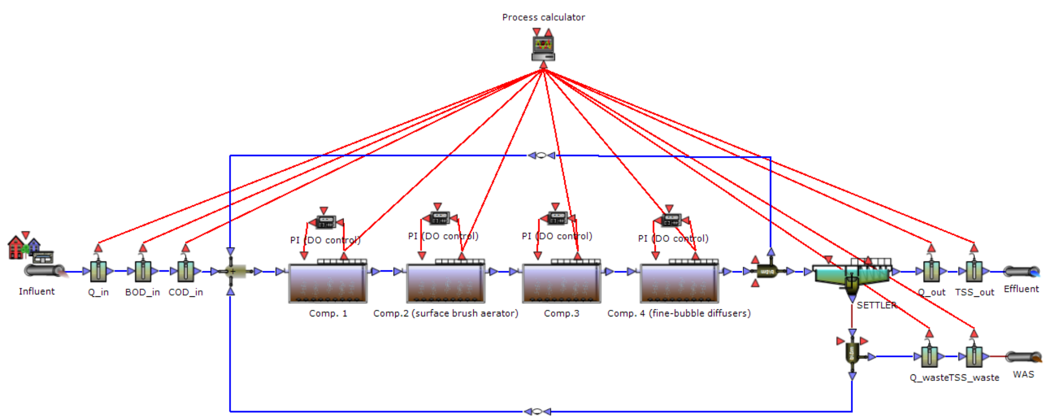


Figure 14. The Kirkonkylä WWTP model layout (with sensors and calculator) in the WEST software

The calculation gave the following values: 43.5 d for SRT, 21 h for HRT and 0.08 $gCOD/gSS$ or 0.03 $gBOD/gSS$ for F/M. The concentrations of MLSS, autotrophic biomass and heterotrophic biomass are at the level of 5,391 g/m^3, 64 g/m^3 and 3,806 g/m^3, when stable nitrification achieved, respectively. However, the real concentrations can differ due to known technical operation problems.

Generally, the calculated operational parameters are close to optimal for oxidation ditch configuration of the ASP. However, F/M of 0.03 $gBOD/gSS$ is lower than recommended value of 0.05 to 0.15 $gBOD/gSS$ and MLSS of 5,391 g/m^3 is higher than recommended value of 1,500 to 5,000 g/m^3 (Metcalf & Eddy 1991).

The stable nitrification is achieved at 250–280 days (Appendix 3), which is extended. It indicates that reactions in the ASR are slow due to low substrate concentration and low temperature (yearly average $T_{eff} = 11.7\ ℃$).

8.2.2 Scenario No. 1: Potential treatment performance

The scenario was performed to define the potential treatment efficiency at the Kirkonkylä WWTP. It was assumed that the nitrification was worsened because of low substrate concentration due to infiltration/exfiltration. This fact was tuned in calibration adjusting K_{NH} parameter (from 1 to 9 $gNH_3 - N/m^3$).

In this scenario, the value of the kinetic parameter K_{NH} was set to the default value of 1 $gNH_3 - N/m^3$ to simulate nitrification in normal operating conditions. The comparison of NH_4, NO_3 and TN content in effluent with and without infiltration/exfiltration is illustrated in Figure 15.

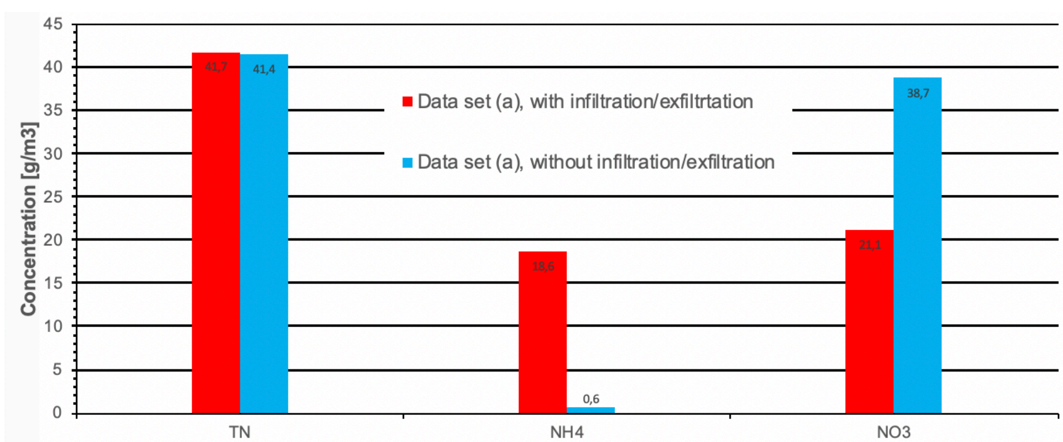


Figure 15. Comparison of NH_4, NO_3, TN with and without infiltration/exfiltration

When examining the simulation results, it can be seen that the high efficiency of NH_4 treatment can be achieved during a stable nitrification with sufficient substrate concentration. NH_4 turns into NO_3 during nitrification, and TN stays nearly at the same level because no anoxic condition for denitrification presents in the system. Thus, better TN removal can be achieved in case of NO_3 utilisation by addition of denitrification in the process. The results of Scenario no. 1 in the WEST software are shown in Appendix 3.

8.2.3 Scenario No. 2: Operation under increased hydraulic load

Another simulated scenario was performed to forecast the treatment efficiency under extremely high hydraulic load compared to the designed hydraulic load of 300 m^3/d. The scenario was simulated to test the Kirkonkylä WWTP performance in perspective.

The data set (a) was used with the same organic load and fractionation of state variables but with increased wastewater flow to 612 m^3/d representing the sum of the yearly average hydraulic load at the Kirkonkylä, Siviilipalveluskeskus and Porlammi WWTPs. The current state of the facility and operational settings were considered and the increase in infiltration/exfiltration overtime was not taken into account. The results awere compared with the results of the data set (a) values under the observed hydraulic load (Figure 16).

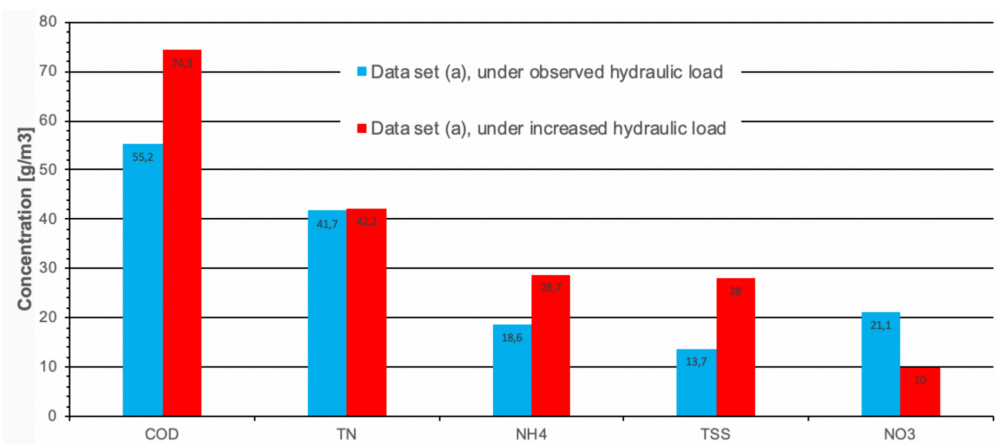


Figure 16. Operation under observed and increased hydraulic load

As it was reported in the wastewater treatment plant annual report 2018 (Ramboll), the Kirkonkylä WWTP can operate at a higher load compared to the design values. The obtained simulation results partially confirmed this fact. The plant treatment performance at a hydraulic load of more than two times higher of the designed showed good results with regard to the nitrogen treatment. However, with regard to the organic matter, TSS and NH_4 removal, the plant is not able to operate efficiently. The results of Scenario no. 2 in the WEST software are demonstrated in Appendix 3.

8.3 Process optimisation suggestions

Analysing the technological process and the current state of the Kirkonkylä WWTP the following suggestions on optimisation can be discussed: adjustment of F/M and SRT by changing RAS and WAS flow rates, the addition of the anoxic tank upstream of oxidation ditch, and aeration optimisation.

Firstly, SRT and F/M have fundamental importance in the design and control of the ASP (Henze et al. 2011). The calculated value for SRT is in the upper bound of the recommended value, and F/M of 0.03 $gBOD/gSS$ is lower than the recommended value of 0.05 to 0.15 $gBOD/gSS$ with the current operation setting. SRT can be increase by increasing RAS or decreasing WAS flow rates. However, this adjustment will lead to the higher nitrification velocity, which is negligible at the operation temperature of 10 ℃ (Shammas 1986). Subsequently, it will decrease NH_4 and NO_3 ration in the effluent, but not TN content because of no anoxic zone for adequate denitrification presents. Besides, an additional source of carbon can be added to increase F/M ration which will positively affect NH_4 removal.

Secondly, the treatment process can be modified to achieve partial denitrification. The plant layout can be modified to the Modified Ludzack-Ettinger (MLE) process by adding anoxic tank upstream of the oxidation ditch along with RAS recirculation. It will enhance the nitrogen removal by achieving partial denitrification. However, denitrification velocity can be low because of a low operating temperature.

Finally, a modification associated with an aeration optimisation can be done to achieve simultaneous nitrification and denitrification inside the oxidation ditch. It is a relatively new and effective method that provides enhancing nitrogen removal at a low level of oxygen concentration. In contrast, more site research and expertise are needed to fine-tune the aeration equipment to reach simultaneous nitrification and denitrification.

9 DISCUSSION

The possibility of setting up the model of small-scale WWTP based only on routine operational and design data has been shown in practice. The working model of the Kirkonkylä WWTP has been built in the WEST software. Nevertheless, outputs of the simulations should not be considered as benchmarks because some assumptions about the plant operation have been made due to the lack of data.

Initially, the scope of the modelling was the organic matter and nitrogen removal prediction. The achieved simulation results for the nitrogen removal have a sufficiently high accuracy. In opposite, accurate simulation predicting the organic matter removal was not achieved due to known plant operation faults that would not be modelled in the ASM1. Moreover, the processes such as infiltration/exfiltration, foaming, bulking, filamentation and deflocculation presenting in the plant operation process, are not described in the ASM family and not considered and extensions are required to include it in the model.

Even though only steady-state simulations were used, several application cases were identified. The following potential application of ASP modelling at a small-scale WWTP has been found: process understanding, prediction of future operation, optimisation and design. Fundamentally, the application cases are related to long-term behaviour investigation without inherent dynamics.

As for the Kirkonkylä WWTP, according to the simulations results, the potential treatment efficiency is not fully achieved due to ageing. The Kirkonkylä WWTP designed capacity of 300 m^3 is not a limit for the hydraulic load, but with the rising hydraulic load, the treatment efficiency falls significantly. The possible optimization of the current ASP modification, i.e. oxidation ditch, is mainly limited due to low temperatures. The average operation temperature is relatively low for successful nitrification and denitrification, especially during the cold season.

10 CONCLUSION

To sum up the main results of the study, the mathematical modelling of the ASP is a valuable tool that can provide a wide range of possibilities for the design, optimisation, research and control of the wastewater treatment process. The application of the models at small-scale WWTPs is usually strictly limited due to an inadequate monitoring approach. Nevertheless, the modelling of a plant structure is feasible. The modelling procedure is manageable, and the obtained model can be used to designed WWTP as well as to likewise constructed WWTPs.

The modelling procedure should be organised in a steplike manner; the most complicated stages in the modelling of small-scale WWTPs are data collection and reconciliation, and calibration and validation. Multiple uncertainties and faults can show up completing these modelling steps. However, a strict adherence to expertise and step-by-step guidelines will help in overcoming obstacles and achieving favourable modelling results.

The decisions made in the design and optimisation processes have a significant financial and environmental impact. Therefore, they must be based on high-quality models. Meanwhile, detailed data on the plant operations and wastewater characterisation are critical for high-quality models. Moreover, the full potential of modelling with accurate outputs can be unleashed only when complete data is available since small systems are highly sensitive to any fluctuations.

To sum up, modelling for optimisation and operation at small-scale WWTPs should be used with caution as many unknown factors due to the limited data can affect simulated outputs. In contrast, modelling can be a valuable tool for the design purpose during the planning and construction phase of small-scale WWTPs to consider future scaling and operational settings in real-time.

11 RECOMMENDATIONS

It is recommended that the Kirkonkylä WWTP should be restored to enhance to overall efficiency as operational problems are identified. In opposite, the restoration of the current WWTP will not resolve the issue with the scaling because the plant already operates over its designed capacity. Nevertheless, some minor actions can be done to improve the situation before a new WWTP is developed and put into operation.

The upgrade of operating WWTPs is not considered reasonable as much expertise and fieldwork are needed, which will be costly and will not provide the future base for the wastewater treatment in the municipality. It is sufficient to construct a new WWTP that will combine in one all operating WWTPs, the Kirkonkylä, Siviilipalveluskeskus, and Porlammi, in the Lapinjärvi municipality.

It was studied that the ASP modelling of a small-scale WWTP can be successfully applied for the technological process study, identification of operational problems and forecasting of future scenarios. Generally, the modelling application is associated with long-term predictions which are sufficient for use in the design of a new WWTP. Briefly, it is assumed that modelling can find a proper practical application in the development of a new WWTP in the Lapinjärvi municipality.

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LISTS OF FIGURES

LIST OF TABLES

The ASM1 process, kinetics and stoichiometry for carbon oxidation, nitrification, and denitrification (the Gujer-Petersen matrix)

Most biochemical processes occur simultaneously in the ASP. The Gujer-Petersen matrix is standard widely employed for their description in ASM1 and others. The matrix notation incorporating carbon oxidation, nitrification, and denitrification processes is introduced in Table 1.

The matrix has columns (i) 1–13 with variables "c_i" and rows (j) 1–8 with processes "ρ_j" where they are involved in. The intersection of i and j in the table gives entries "v_{ij}" defining the process rate for variables. According to the Gujer-Petersen notation:

- $v_{ij} = 0$ (empty) if c_i is not affected by ρ_j
- $v_{ij} < 0$ if c_i is a substrate of ρ_j
- $v_{ij} < 0$ if c_i is a product of ρ_j

v_{ij} may be adimensional if all state variables are expressed according to the same measuring unit, or they may be dimensional if hybrid units are used.

The net reaction rate of a variable "r_i" is the sum of all the process rates, which cause a change in the mass of the variable. It is introduced in the last column for each process.

$$r_i = \sum_j v_{ij}\rho_j$$

Process kinetics and stoichiometry for carbon oxidation, nitrification and denitrification (Henze et al. 1987)

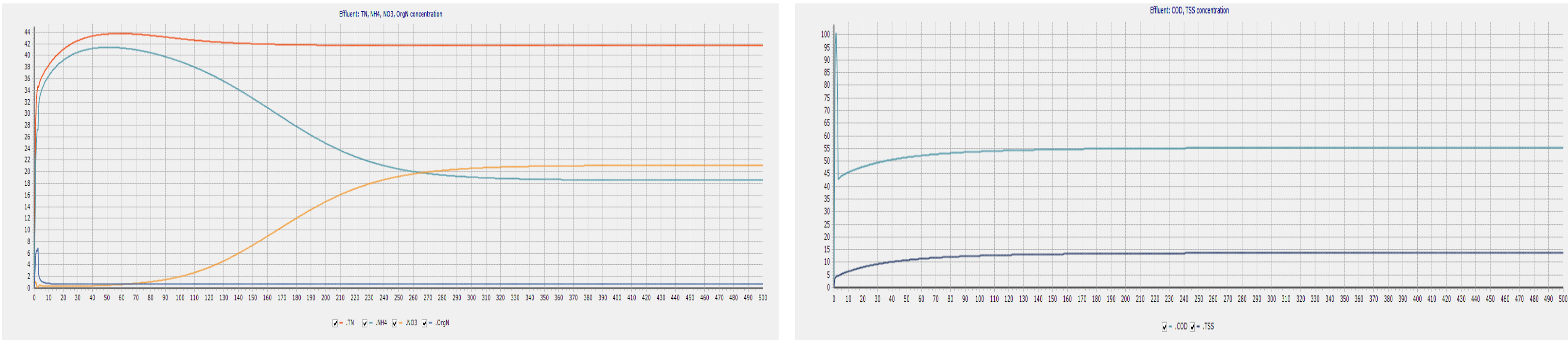
Variable →	i	1	2	3	4	5	6	7	8	9	10	11	12	13	Process Rate, ρ_j $[ML^{-3}T^{-1}]$
j	Process ↓	S_I	S_S	X_I	X_S	X_{BH}	X_{BA}	X_P	S_O	S_{NO}	S_{NH}	S_{ND}	X_{ND}	S_{ALK}	
1	Aerobic growth of heterotrophs		$-\frac{1}{Y_H}$			1			$-\frac{1-Y_H}{Y_H}$		$-i_{XB}$			$-\frac{i_{XB}}{14}$	$\mu_H\left(\frac{S_S}{K_S+S_S}\right)\left(\frac{S_O}{K_{OH}+S_O}\right)X_{BH}$
2	Anoxic growth of heterotrophs		$-\frac{1}{Y_H}$			1				$-\frac{1-Y_H}{2.86Y_H}$	$-i_{XB}$			$\frac{1-Y_H}{14\times 2.86Y_H}-\frac{i_{XB}}{14}$	$\mu_H\left(\frac{S_S}{K_S+S_S}\right)\left(\frac{K_{OH}}{K_{OH}+S_O}\right)\left(\frac{S_{NO}}{K_{NO}+S_{NO}}\right)\eta_g X_{BH}$
3	Aerobic growth of autotrophs						1		$-\frac{4.57-Y_A}{Y_A}$	$\frac{1}{Y_A}$	$-i_{XB}-\frac{1}{Y_A}$			$-\frac{i_{XB}}{14}-\frac{1}{7Y_A}$	$\mu_A\left(\frac{S_{NH}}{K_{NH}+S_{NH}}\right)\left(\frac{S_O}{K_{OA}+S_O}\right)X_{BA}$
4	Decay of heterotrophs				$1-f_P$	-1		f_P					$i_{XB}-f_P i_{XP}$		$b_H X_{BH}$
5	Decay of autotrophs				$1-f_P$		-1	f_P					$i_{XB}-f_P i_{XP}$		$b_A X_{BA}$
6	Ammonification of soluble organic nitrogen										1	-1		$\frac{1}{14}$	$k_a S_{ND} X_{BH}$
7	Hydrolysis of entrapped organics		1		-1										$k_h\frac{X_S\div X_{BH}}{K_X+(X_S\div X_{BH})}\left[\left(\frac{S_O}{K_{OH}+S_O}\right)+\eta_h\left(\frac{K_{OH}}{K_{OH}+S_O}\right)\left(\frac{S_{NO}}{K_{NO}+S_{NO}}\right)\right]X_{BH}$
8	Hydrolysis of entrapped organic nitrogen											1	-1		$\rho_7(X_{ND}\div X_S)$
Observed Conversion Rates $[ML^{-3}T^{-1}]$		$r_i=\sum_j v_{ij}\rho_j$													
Stoichiometric Parameters: Heterotrophic yield: Y_H Autotrophic yield: Y_A Fraction of biomass yielding particulate products: f_P Mass N / Mass COD in biomass: i_{XB} Mass N / Mass COD in products from biomass: i_{XP}		Soluble inert organic matter $[M(COD)L^{-3}]$	Readily bLiodegradable substrate $[M(COD)L^{-3}]$	Particulate inert organic matter $[M(COD)L^{-3}]$	Slowly biodegradable substrate $[M(COD)L^{-3}]$	Active autotrophic biomass $[M(COD)L^{-3}]$	Active autotrophic biomass $[M(COD)L^{-3}]$	Particulate products arising from biomass decay $[M(COD)L^{-3}]$	Oxygen (negative COD) $[M(-COD)L^{-3}]$	Nitrate and nitrite nitrogen $[M(N)L^{-3}]$	NH_4^++NH_3 nitrogen $[M(N)L^{-3}]$	Soluble biodegradable organic nitrogen $[M(N)L^{-3}]$	Particulate biodegradable organic nitrogen $[M(N)L^{-3}]$	Alkalinity $[Molar\ units]$	*Kinetic Parameters:* Heterotrophic growth and decay: μ_H, K_S, K_{OH}, K_{NO}, b_H Autotrophic growth and decay: μ_A, K_{NH}, K_{OA}, b_A Correction factor for anoxic growth of heterotrophs: η_g Ammonification: k_a Hydrolysis: k_h, K_X Correction factor for anoxic hydrolysis: η_h

Stoichiometric and kinetic parameters in the ASM1 model

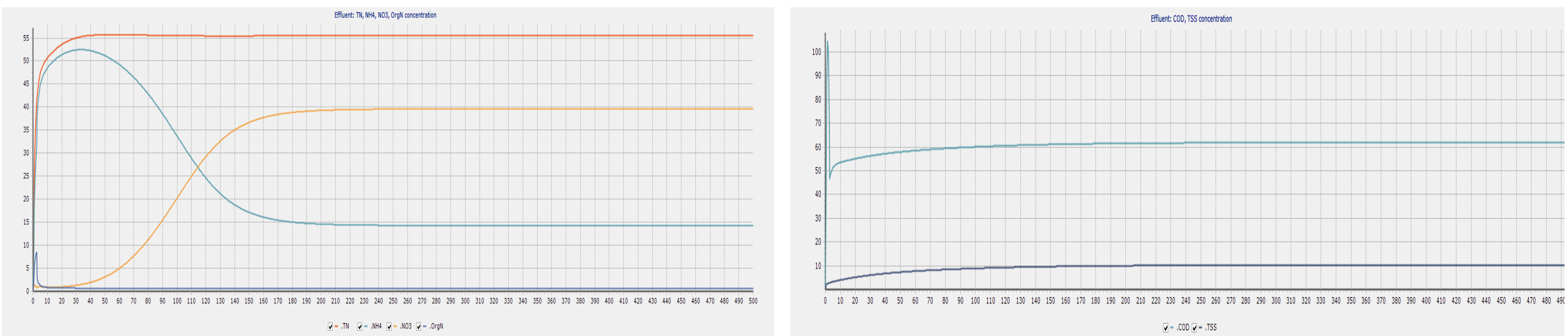
Typical parameter values at neutral pH (Henze et al. 1987)

Parameter	**Symbol**	**Unit**	**Value at 20 ℃**	**Value at 10 ℃**
Stoichiometric parameters				
Heterotrophic Yield	Y_H	$g(cellCOD\ formed)/g(COD\ oxidized)$	0.67	0.67
Autotrophic Yield	Y_A	$g(cellCOD\ formed)/g(N\ oxidized)$	0.24	0.24
Fraction of biomass yielding particulate products	f_P	$dimensionless$	0.08	0.08
(Mass N) / (Mass COD) in biomass	i_{XB}	$gN/gCOD$	0.086	0.086
(Mass N) / (Mass COD) products from biomass	i_{XP}	$gN/gCOD$	0.06	0.06
Kinetic parameters				
Heterotrophic maximum specific growth rate	μ_H	$1/d$	6.0	3.0
Hsc for heterotrophs	K_{SH}	$gCOD/m^3$	20.0	20.0
Oxygen hsc for heterotrophs	K_{OH}	gO_2/m^3	0.20	0.20
Nitrate hsc for heterotrophs	K_{NO}	$gNO_3 - N/m^3$	0.50	0.50
Heterotrophic decay rate	b_H	$1/d$	0.62	0.20
Correction factor for growth for heterotrophs	η_g	$dimensionless$	0.80	0.80
Autotrophic maximum specific growth rate	μ_A	$1/d$	0.80	0.30
Ammonia hsc for autotrophs	K_{NH}	$gNH_3 - N/m^3$	1.0	1.0
Oxygen hsc for autotrophs	K_{OA}	gO_2/m^3	0.40	0.40
Autotrophic decay rate	b_A	$1/d$	0.15	0.10
Ammonification rate	k_a	$m^3/gCOD/d$	0.08	0.04
Maximum specific hydrolysis rate	k_h	$g(slowlybiodegr.COD)/g(cellCOD)/d$	3.0	1.0
Hsc for hydrolysis of slowly biodegradable substrate	K_X	$g(slowlybiodegr.COD)/g(cellCOD)$	0.03	0.01
Correction factor for anoxic hydrolysis	η_h	$dimensionless$	0.40	0.40

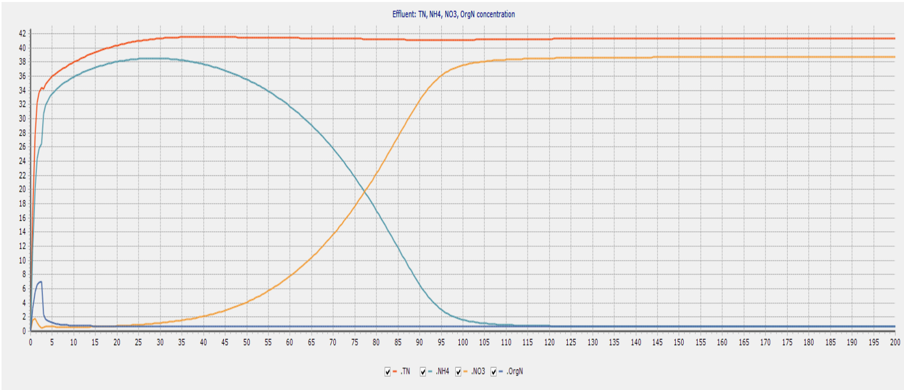
Simulation results in the WEST software

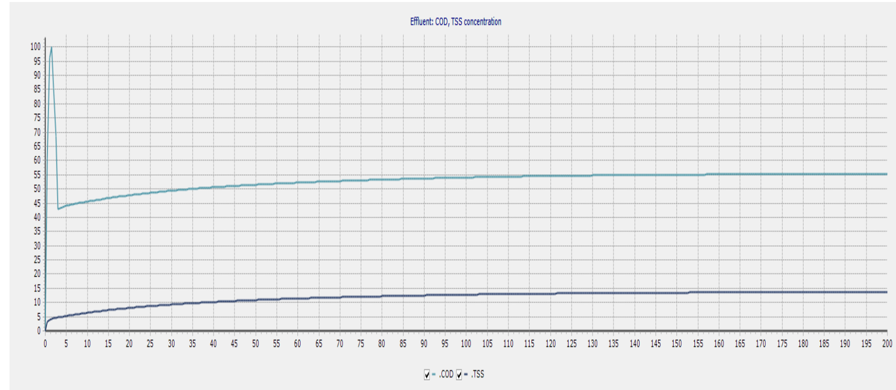


Calibration: Steady-state simulation results of the calibrated data set (a)

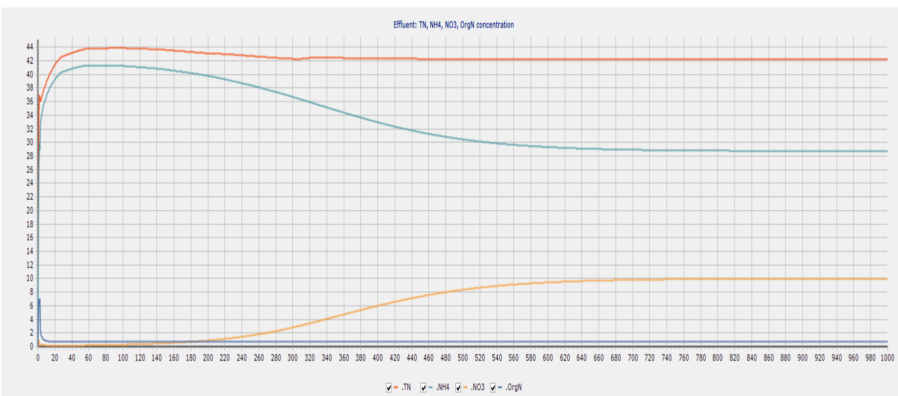


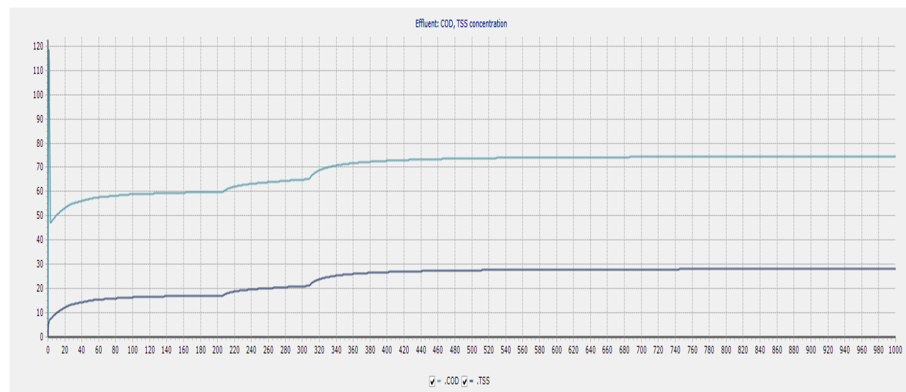
Validation: Steady-state simulation results of the calibrated data set (b)





Scenario No. 1: Steady-state simulation results of the calibrated data set (a) without infiltration/exfiltration





Scenario No. 2: Steady-state simulation results of the calibrates data set (a) under increased hydraulic load of 612 m^3/d