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COD fraction dynamics: Respirometric analysis & modelling sewer processes

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DECLARATION

I declare that the work presented in this thesis is, so to the best of my knowledge and belief, original and my own work, except as acknowledged in the text.

Berlin, 14/08/2007 Christian Köhler

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Engineering is not merely knowing and being knowledgeable, like a walking encyclopaedia; engineering is not merely analysis; engineering is not merely the possession of the capacity to get elegant solutions to non-existent engineering problems; engineering is practicing the art of the organized forcing of technological change ... Engineers operate at the interface between science and society ...

Dean Gordon Brown

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List of symbols and abbreviations

$\mu_{max,A}$	Maximum specific growth rate of autotrophic biomass [d^{-1}]
ASM	Activated Sludge Model (Henze et al. (2000))
ATU	Allythiorea
COD	Chemical Oxygen Demand
CSO	Combined Sewer Overflow
CSTR	Continuous Stirred-Tank Reactor
DO	Dissolved Oxygen [mgL^{-1}]
DWF	Dry weather Flow
EU WFD	European Water Framework Directive
k_h	Maximum specific hydrolysis rate [d^{-1}]
K_La	Aeration capacity [h^{-1}]
MSL	Model Specific Language
ODE	Ordinary Differential Equations
PAO	Polyphosphate Accumulating Organism
PE	Population Equivalent
RTC	Real-Time Control
S_I	Inter, (soluble), non-biodegradable organics [$mgCODL^{-1}$]
S_S	Readily, (soluble), biodegradable organics [$mgCODL^{-1}$]
TKN	Total Kjeldahl Nitrogen [mgL^{-1}]
TP	Total Phosphorus [mgL^{-1}]
TSS	Total Suspended Solids [kgm^{-3}]
TVS	Total Volatile Solids [kgm^{-3}]
VSS	Volatile Suspended Solids [kgm^{-3}]
WWTP	Wastewater Treatment Plant
X_H	Heterotrophic biomass [$mgCODL^{-1}$]

X_I inter, (particulate), non-biodegradable organics [$mgCODL^{-1}$]

X_S slowly, (particulate), biodegradable organics [$mgCODL^{-1}$]

1 INTRODUCTION

For hundreds of years wastewater management has been developed and improved to match the dynamic needs of man and nature.
Methods for sewage disposal already exist since antiquity. Often people built drainage systems to handle storm runoff. The Romans constructed drainage systems that were particularly sophisticated since they also drained wastewater. A part of them is still in use today. At the end of the middle ages subterranean collecting pits arose in European basement vaults. Later on cesspools were built whereas these collecting basins were rather an exception. Usually, the sewage was emptied on the streets and was flushed away by rainwater afterwards. With growing population in bigger cities the risk of disastrous epidemic outbreaks became very high. Due to ineffective separation of sewage and drinking water many habitants suffered from cholera or typhoid fever. In the middle of the 19th century the first combined sewer system was constructed in London. The sewage was directly conveyed into the river Thames. By and by people recognized that the direct discharge of sewage into the river produces an excessive load burden on the water body that leads to significant influences on flora and fauna. Furthermore, polluted water caused serious health damage. Thus, in the beginning of the 20th century wastewater treatment plants were developed which treated the sewage by percolating filter processes. In 1910 the first activated sludge process was applied to treat wastewater. This process constituted a milestone in wastewater treatment and resulted in a considerable improvement of the receiving water quality and was further developed until today.

Currently, the urban wastewater management is being confronted with always new challenges that focus on design, optimisation and control of sewer and treatment systems to satisfy the increasing requirements of water quality. Often, improvements can be achieved by modernisation of already existing infrastructures in industrial countries but in Third World and threshold countries treatment systems still have to be developed in an integrated and sustainable way.
Furthermore, a rethinking in design and optimisation of wastewater structures has taken place in the western world due to the rigorous legislation that is being enforced like the European Water Framework Directive passed in 2000. It implies a holistic approach of all urban systems (including the receiving water body) related to wastewater to ensure a sustainable urban wastewater management that is required to achieve an extensive water quality improvement.
Moreover, new challenges for engineers arise from climate change. In February 2007 the IPCC (Intergovernmental Panel on Climate Change) accepted and adopted the first part of its fourth assessment report on climate change. Within this report the latest science revealed unequivocal

evidence that the mean temperature is rising. Resulting impacts are already present and will intensify. From a water perspective, the key issues are more extreme weather events and the likelihood of more floods and droughts. More droughts would mean a greater necessity to ensure that the little portable water that is available is of adequate quality. More floods would imply that the urban wastewater system must be resilient enough to assimilate the pollution and hydraulic loads not of the fifth or tenth year typical storm of pipe design but of the more extreme storm events which can already be observed under climate change and are likely to worsen in the future.
All the above pressures will necessitate more and more research and development of urban wastewater systems.

A relatively new tool to pursue the goals of water quality improvement and maintenance are software packages that are being used to asses, optimize or design wastewater systems. Models are thereby capable of replicating interactions between different subsystems of the urban wastewater system in an integrated way as well as to predict water flow and quality. These models are continuously being improved with the aim of better accuracy and lower computing time. At this stage one can choose among a variety of mathematical models to reproduce processes including surface runoff, wastewater flow, treatment and river water quality processes. These models have to be calibrated by using field data to describe the individual characteristic of each subsystem.

The investigations within this thesis (Diplomarbeit) were conducted at the research group model*EAU* based at Université Laval (Québec, Canada) and deal in general with modelling and optimising integrated urban wastewater systems. It is related to the Ph.D. study of Anne-Marie Solvi ("Modelling the Sewer-Treatment-Urban River System in view of the EU Water Framework Directive"; Solvi (2007)) that focused on integrated urban wastewater systems. The main aim of Solvi (2007) consisted of implementing of the KOSIM model, which is a hydrological based sewer network model (ITWH (2000)), into the simulation software WEST® developed at the Biomath department of Ghent University (Belgium) in collaboration with MOSTforWATER N.V. (Kortrijk, Belgium) to produce an integrated replication for the Bleesbruck catchment in Luxembourg.
In particular, the Diplomarbeit deals with analysis of the sewer model and the related prediction of the soluble and particulate biodegradable substrate concentration in the wastewater treatment plant (WWTP) inlet. A large part of this work comprises laboratory analysis of the COD (Chemical Oxygen Demand) fractionation needed in an Activated Sludge Model (ASM). Wastewater from the inlet of the Saint Nicolas treatment plant (Québec, Canada) was analysed by using respirometry tests in order to evaluate the effect of processes represented in the sewer model on wastewater

composition at the WWTP inlet. The WEST® software is used to investigate process and operation scenarios of the Bleesbruck sewer model with regard to their influence on COD fractionation with a special focus on readily and slowly biodegradable substrate. The aim is to provide a basis for improving the accuracy of the WWTP inlet fractionation for ASM modelling.

The present thesis is divided into four main parts. The first part will give the reader an introduction to sewer and wastewater treatment plant modelling in general. Afterwards, descriptions of some case studies show how modelling was applied to provide decision support for improving the integrated urban wastewater system. The third part comprises respirometric analysis of Quebec wastewater where assumptions are made regarding the diurnal fluctuation of the COD fractions. These assumptions will be further investigated by using and modifying the Bleesbruck sewer model in the fourth part of this work.

2 MODELLING APPROACH OF THE SEWER SYSTEM AND THE WWTP

2.1 INCENTIVE FOR MODELLING OF THE INTEGRATED URBAN WASTEWATER SYSTEM

2.1.1 THE EUROPEAN WATER FRAMEWORK DIRECTIVE

The need for wastewater system modelling increases because of the changing approach on urban water management. On the 22nd December in 2000 the legislation of the European Union passed a water protection directive based on an integrated urban water management that is based on a combination of two competing principles: emission and immission. The former defines a minimum standard for wastewater treatment, i.e. the emission limit values principle aims at applying best available technologies to ensure the lowest pollutant discharge possible and may be considered as rather "environment orientated". The latter principle defines quality standards for water bodies (important for protection). The "water quality objectives" also called immission limit values approach) take into account the natural capacity of water at the point of discharge when setting standards. It may be considered as more public health orientated.

The Water Framework Directive (WFD) aims to spread and define the European regulations for water protection. It takes a holistic approach to water management, as water flows through a catchment from lakes, rivers and groundwater towards estuaries and thence the sea. Surface and groundwater are considered together, in both qualitative and quantitative terms. It is required to achieve a “good surface water status” and a “good groundwater status” as well as to prevent deterioration in water quality of those waters which are already “good” in terms of chemical and ecological quality (Griffiths (2002)).

The starting point is the protection of all waters (not only waters with a certain use), surface waters and groundwater by managing water quality and quantity. The including priority objective is the achievement of a good water condition by 2015. For groundwater this means both quality in the physicochemical sense and sufficient quantity balanced between groundwater withdrawals and enrichment. For surface waters it implies the aim of a good condition with respect to physicochemical and ecological quality. A good condition of the surface water requires at least “good” ecological and chemical conditions whereas a good condition of the groundwater describes at least “good” quantitative and chemical conditions. Good chemical conditions means that no pollutant has a higher concentration than defined in the environmental quality standards.

The European Water Framework Directive also includes several principles. Among others it contains the directive on treatment of municipal wastewater where a decrease of surface water pollution, in particularly due to nutrients, from waste water is regulated. According to this, catchment sewerage and secondary treatment of the wastewater are compulsory for more than 2000 PE (population equivalents). The secondary treatment comprises carbon removal in order to reduce the organic substrate load discharged in the water body. Moreover, the directive implies an advanced treatment comprising phosphorus and nitrogen removal in "sensitive areas". These areas are water catchments used for drinking water production purposes or with upcoming eutrophication.

According to the rules of the EU WFD, many countries made more rigorous regulations to assess and improve the status of surface- and groundwater. The given requirements of water quality imply higher standards in design and control as well as diagnosis and optimization of wastewater systems. The integrated approach requires a large amount of data to assess the system. Hence, models are used in computer simulations to support the high standard of water management. Today a lot of software packages (INFOWORKS, KOSIM,MIKE, MOUSE, SIMBA, SIMULINK, WEST®, etc.) are available which include different models to predict situations in the integrated urban water system and give answers to precise questions. Thereby, the choice of a model mostly depends on what should be considered and on the purpose to be achieved. These factors determine the amount of data being necessary to get an appropriate description of reality. In some cases the aim may have to be adapted to the possibilities of an available model. The implemented models include a variety of different submodels. Their complexity is of particular importance with respect to the limited computing time and available data. In general it can be stated that the model is supposed to be as detailed as necessary but as simple as possible (Muschalla (2006)).

2.1.2 The integrated urban wastewater system

To have a closer view at the structure of integrated modelling the system has to be divided into the main components of the integrated urban wastewater system: catchment, sewer system, wastewater treatment plant (WWTP) and river. The integrated urban wastewater system constitutes a part of the global water system and is in interaction with other parts like groundwater and seas. However, the wastewater system considers urban settlements and its influence on water quality of the receiving water body located in the neighbourhood of the city or village.

In the following a brief introduction to the influence of each subsystem on water quantity and quality will be given according to the flow path the water takes until it arrives in the water body (See Figure 1).

Rain water reaches the urban catchment where it is transformed into runoff. The quantity of that water is certainly dependent on the rain intensity. However, losses have also to be taken into account that are caused by evaporation, wetting and depression. Furthermore, the ratio of impervious / pervious area influences the developing runoff. At the drained catchment the water and the wastewater enter into the combined sewer system together whereas the wastewater is originating domestically and industrially. This kind of sewer system is the most common method to drain larger cities. Rain water running from agricultural land can carry significant loads of phosphorus and nitrogen because these chemicals are used as fertilizer and impact the water quality. One considers agricultural land as diffuse pollutant sources for water bodies. The sewer system transports the wastewater to the treatment plant by gravity or via pressure pipes. Along the way it is subject to sedimentation, erosion and biological processes that will be focused upon later in this work. Stormwater tanks installed in the sewer system relieve the WWTP during rain events with respect to water volume and (depending on the construction of the tank) pollutants. During heavy rain it is possible that the capacity of the retention basin is reached and a part of the wastewater spills over into the river which is the significantly influenced by such an overflow event. The wastewater treatment plant purifies the sewage by using physical, chemical and biological processes and, afterwards, the treated water is discharged into the water body. Certain threshold values have to be met at the final effluent discharge point.

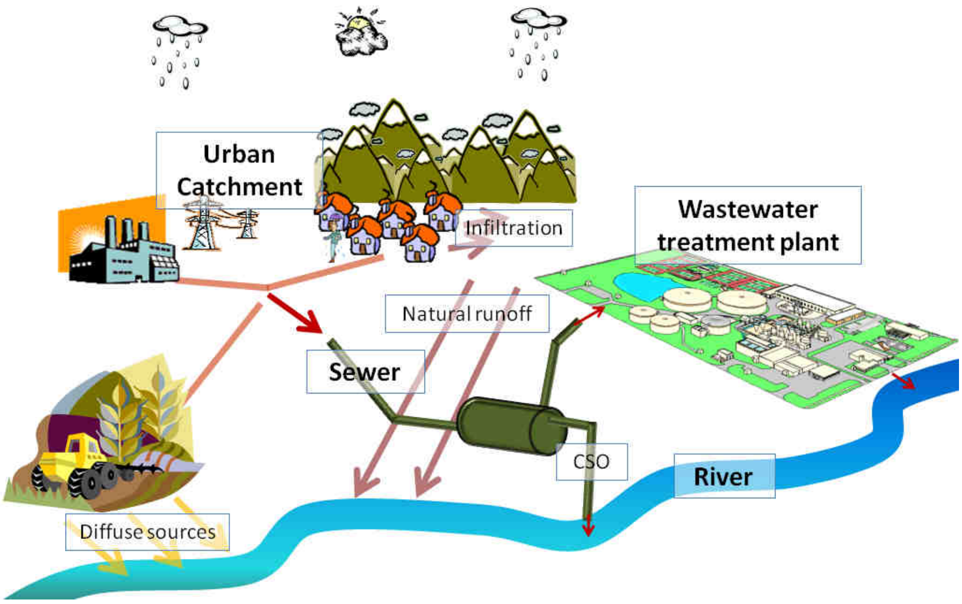


FIGURE 1 ILLUSTRATION OF THE URBAN WASTEWATER SYSTEM. THE WASTEWATER TREATMENT PLANT AND ESPECIALLY THE SEWER SYSTEM WILL BE STUDIED WITHIN THIS THESIS.

According to Figure 1 it can be shown that the different subsystems of the urban wastewater system are related to each other. Thus, for improving the river water quality a holistic approach has to be followed when the system or only a part of it is to be evaluated and optimized. To give a trivial example: by only modernising the treatment processes the desired extent of river water quality improvement could not be achieved if sewer overflow events are not taken into consideration.
Within this thesis focus will be on the wastewater treatment plant and especially the sewer system which comprises the urban catchment.

2.2 Sewer Modelling

2.2.1 Sewer catchment

The sewer catchment describes a delimited area where rainwater runs off from the surface and is collected by the sewer system. To model the rainfall-runoff process a lot of hydrological data is required to determine the losses occurring between the arriving water on the surface and the collection in the sewer system. According to the extent of impermeability the area is divided into different classes in order to determine the infiltration losses. Hereby, one should also consider depression losses and wetting of the surface. The climate in which the considered catchment is situated has a distinctive influence on the evapotranspiration which presents another loss in the mass balance of the runoff water. Overland flow process modelling ranges from very simple (reservoir models, time-area relations or unit hydrographs) to detailed models like the kinematic-wave model (Butler and Davies (2000)).

2.2.2 Sewer flow modelling

The surface runoff hydrograph constitutes an important input for the sewer system model.
Clearly defined parameters regarding pipes, structure and geometry are required to achieve good results from the *hydraulic model* of the sewer system. The water movement presented by physical processes (like unsteady flow conditions in open channels) can be determined by the “de Saint-Venant” equations. These two equations describe the conservation of mass (continuity equation (1)) and the conservation of momentum (dynamic equation (2)) (Butler and Davies (2000))

$$B\frac{\partial y}{\partial t}+\frac{\partial Q}{\partial x}=0$$ (Continuity equation) (1)

$$\frac{\partial Q}{\partial t}+\frac{\partial}{\partial t}\left(\frac{Q^2}{A}\right)+gA\frac{\partial y}{\partial x}-gA\left(S_0-S_f\right)=0$$ (Dynamic equation) (2)

Where: y .. Flow depth or water level [m]

Q .. Flow rate [m³/s]

A .. Area of flow cross-section [m²]

B .. Water surface width [m]

t .. Time [s]

x .. Distance [m]

S_0 .. Bed slope [-]

S_f .. Friction slope [-]

Depending on the purpose some of the terms might be less significant so that simplifications can be made. The so-called dynamic wave equations imply wave attenuation, backwater effects as well as flow acceleration. The Saint-Venant equations are function of time (t) and distance (x) so that numerical methods (finite elements or finite differences) are needed to solve the set of equations numerically. The problem of inaccuracy can only be met by using little time steps that negatively influence the computing time. Thus an appropriate compromise between accuracy and resulting calculating time of the model must be found. To reproduce a good (virtual) picture of the water movement in the sewer system interfaces are used to feed the software packages (i.e. the delivered hydraulic equations) with information (like connections, dimensions, slopes and locations) to take manholes, overflows and weirs into account (Meirlaen (2002)).

Besides the hydrodynamic models there are also *hydrological models* to reproduce the sewer system where pipes are modelled as “black-box”. Hereby, no exact physical processes are considered. In fact, a transfer function is implemented to describe the water movement. One approximates the unsteady flow in a sewer pipe by splitting it up into different stretches, in each of which a steady flow is occurring. Consequently, the continuity and the momentum equation are replaced by a retention function and a Flow-Volume relationship (see equation (3) and (4); Solvi (2007)).

$$\frac{\partial V}{\partial t} = Q_{in}(t) - Q_{out}(t) \tag{3}$$

$$Q_{Out}(t) = \frac{1}{k} V(t) \tag{4}$$

Where: Q_{in} .. Inflow [m^3]

Q_{out} .. Outflow [m^3]

V .. Volume in a certain stretch [m^3]

k .. Residence time [s]

Compared to the hydrodynamic models the results of the hydrological approach are less exact and typically do not consider special situations like backwater flow or pressurized flows. Yet, convincing advantages exist, e.g. the notably lower computing times and the higher calculation stability as well as the lower need for data and, hence, the simplicity and transparency of the model structure (see Figure 2).

Despite of the greater impreciseness which is a distinctive weakness, the hydrological approach has proven to be an appropriate tool to test integrated scenarios of the Bleesbruck catchment considered in the thesis (Solvi (2007)).

	Hydrodynamic models	Hydrological models
Accuracy	+	-
Special situations (e.g. pressurized flows)	+	-
Need for data	-	+
Simplicity & transparency	-	+
Computing time	-	+

FIGURE 2 ADVANTAGES AND DISADVANTAGES OF HYDRODYNAMIC AND HYDROLOGICAL MODELS

2.2.3 SEWER QUALITY MODELLING

Besides water movement modelling of the sewer system one must also consider water quality changes (with time) during the transportation process. Therefore, it is required to express those

changes in mathematical equations to implement them into a water quality submodel. The relevant wastewater components (especially solids, ammonia, phosphorus and organic carbon) can be divided in soluble and non-soluble components and are subject to physical, chemical and biological processes taking place in the sewer system. Moreover, the length of dry weather periods, the magnitude of sudden storm events and therewith first flush effects (Krebs et al. (1999), Huisman et al. (2000)) have a notable impact on processes in the sewer pipe. The latter can also be considered as a bioreactor (Hvitved-Jacobsen et al. (2002)) where non-negligible changes in pollutant concentration occur before the sewage reaches the treatment plant. Size and construction of the sewer system thereby have a considerable influence on biodegradation that benefits from sufficient reaction and flow time and sufficiently high oxygen concentrations (and/or nitrate concentrations under anoxic conditions) for respiration (Ahnert et al. (2005)). Thus, one assumes that bio-chemical changes relating to ammonification, hydrolysis, degradation of organic matter and anaerobic digestion are taking place next to physical processes like sedimentation, erosion, resuspension and compaction. Considering all these processes is decisive for proper modelling of an integrated urban water system.
According to Rauch et al. (2002) one can distinguish between 4 main processes:

- pollutant accumulation
- pollutant wash-off
- pollutant transport and transfer
- pollutant conversion processes

Pollutant *accumulation* occurs on the surface and in the sewer where sedimentation of suspended solids is taking place during dry weather periods. This effect results in blockage, loss of hydraulic capacity and pollutant storage. Usually a linear accumulation on an exponentially asymptotic function is used to model the accumulation process.

The accumulated pollutants are *washed off* during storm weather events. Rain is wetting the sediment surface first and afterwards flushes into the sewer. A frequently used model assumes a dependence of the wash-off load and the pollutants which are remaining on the surface, leading to an exponentially decreasing wash-off load (Charbeneau and Barett (1998)). Nevertheless, it should be pointed out that the main accumulation and, therewith, most of the resuspension of pollutants take place in the sewer pipe and is mainly influenced by the first flush effect.
The first flush effect describes the sudden resuspension of suspended and volatile suspended solids into the water phase. With the beginning of a storm event a peak load of pollutants is developed after a certain dry weather period before a dilution effect through the higher rain water flux can take

place. How much material will erode is mostly influenced by temperature, flux, length of the dry weather period and therewith the resulting sediment bed strength. Generally, it can be assumed that a biofilm (covering the sediment surface) develops during the dry weather periods and impacts the erosion properties. However, until today these processes cannot be fully described with mathematical equations. Nevertheless, Ashley et al. (2000) and Skipworth et al. (1999) give an appropriate solution to reproduce the wash-off effect by an equation depending on the accumulated mass of solids, flow velocity or the bed shear stress.

The *transport* of soluble components can be described by reservoir models or mechanistic models (advection-dispersion). It should be pointed out that a first flush effect of dissolved contents is also possible due to the propagation of flood waves at a velocity higher than the flow velocity of the water. At the beginning of a storm event the flux is increased at the end of the pipe by the flood wave without the corresponding decrease in dissolved components. This leads to a peak load (Huisman et al., 2000; Krebs et al., 1999a). The transport of particulates can be divided into suspended load, near bed solids load and bed load. These processes require different considerations and therewith different models (Meirlaen, 2002).

The main *pollutant processes* are stimulated by biofilm processes (aerobic, anoxic and anaerobic conditions can be found). Hvitved-Jacobsen et al. (2002) describe the biological system which is active in the sewer. Huisman (2001) gives a complete biological sewer model based on the Activated Sludge Model No.3 (Gujer et al. (1999)) that also considers the sulphur cycle and gas exchange with the sewer atmosphere. In combination with hydrodynamic equations the model can be seen as a complete reproduction of the sewer system.

2.3 Modelling of biological wastewater treatment plants (WWTP)

2.3.1 Activated sludge models

There are a number of models that are used to simply describe processes occurring in a treatment plant. The most applied approach is dynamic modelling of activated sludge treatment plants. Hence, in the following this kind of modelling will be considered more closely.

A dedicated Task Group of the IWA (International Water Association) developed a model which describes the relevant processes of activated sludge processes (Henze et al. (1987)). This model is based upon a kinetic approach of the metabolism taking into account the conservation of chemical oxygen demand (COD) and nitrogen to deduce the stoichiometry of each process. To reveal all the interactions between the system components in an understandable way the IWA Task Group on mathematical modelling used the matrix format which was developed by Petersen (1965). This matrix contains model components, the stoichiometry of the biological processes and their associated process rates (based on Monod Kinetics) organized in a structured way. The general approach is to find the appropriate growth rate of the microorganisms (autotrophic and heterotrophic) and their effects on the different components. With the introduction of the Activated Sludge Model (ASM) No.1 in 1986 a first “standard” model was published to describe carbon removal and nitrification-denitrification processes. This model has been widely used and also established the basis for further model development leading to ASM No.2 which also includes biological phosphorus removal (see the chronology of ASM development of IWA in Figure 3). The upgraded version ASM No.2d was developed since it has been found out that biological phosphorus removal also occurs under anoxic conditions and polyphosphate-accumulating organisms (PAOs) contribute to the denitrification process (Henze et al. (1999)). The ASM No.2d was used to simulate the Bleesbruck WWTP investigated in this work (Solvi (2007)).

In 1998 ASM No.3 was developed to overcome some defects of the first model which became apparent during the intensive research performed on the basis of ASM No 1.

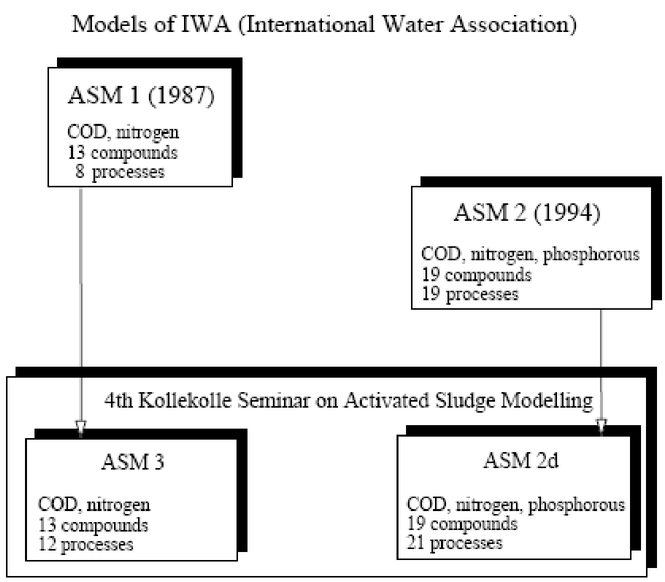


FIGURE 3 ACTIVATED SLUDGE MODEL DEVELOPMENT BY IWA (HTTP:// WWW.IFAK-MD.DE)

As shown in Figure 3 the complexity of the activated sludge models increased with progressive model development whereas all activated sludge models are built upon the basis of the ASM No.1. In the ASM No.1 (Henze et al. (1987)) one can find a parameter list whose eight components do not need to be measured because assumed values are satisfactory. The remaining 20 parameters have to be evaluated according to a specific order since the determination of some parameters assumes that other values are known. The determination is carried out by different methods e.g. examining oxygen uptake rates in batch experiments (Vanrolleghem et al. (1999). By implementation of short or long time respirometric experiments, for example, the readily and slowly biodegradable substrate fraction S_S and X_S in the wastewater can be obtained. The total COD in the influent wastewater is made up of (Henze et al. (1987)):

$$COD = S_S + X_S + X_I + S_I \tag{5}$$

Where: S_S .. readily biodegradable substrate [mg/L]
X_S .. slowly biodegradable substrate [mg/L]
X_I .. inert suspended organic matter [mg/L]
S_I .. inert soluble organic matter [mg/L]

Among the parameters which cannot be determined directly from analytical measurements the heterotrophic yield Y_H, the half-saturation coefficient K_{NH}, the maximum specific growth of autotrophic biomass $\mu_{max,A}$ and the maximum specific hydrolysis rate k_h belong to the most important parameters (Lützner and Kühn (2002)). Those can be estimated by using batch tests or otherwise with the help of mass balance equations applied to pilot plant investigations. After the determination of the parameters and fractions presented in the ASM the validation of the model has to be conducted either at the wastewater treatment plant or through large-scale tests.

2.3.2 Hydraulic models of treatment plants

The hydraulics of the treatment plant are usually modelled in a simplified way by using the variable volume tank approach or using transfer functions (De Clercq et al. (1999)). Generally, the outflow rate is assumed to be equal to the inflow rate at any time. Imperfect mixing in the reactor is presented by the model of continuously stirred tank reactors (CSTR) in series approach (Levenspiel (1972)). The model mimics the advection-dispersion behaviour in different unit processes.

2.3.3 Modelling of clarifiers

Simulation of the clarifier processes (one of the most important unit processes in water treatment) is described by four basic types of settling that take place in the primary and secondary clarification process (Takács et al. (1991)):

- Discrete particle settling (the solids settle as individual entities)
- Flocculent particle settling (typical for primary clarifiers and for the upper layers of secondary settlers)
- Hindered settling (suspension in which inter-particle forces hinder the settling process)
- Compression settling (achieved by compression of the mass of particles)

Most primary clarifier models do not deal with dynamic behaviour. They are rather based on a steady state approach. Primary clarification has to be considered as a fundamental treatment unit because its performance affects subsequent treatment processes and sludge treatment. Lessard and Beck (1988) present a dynamic model including five state variables:

- Total suspended solids (TSS)
- Volatile suspended solids (VSS)
- Chemical oxygen demand (COD)
- Soluble COD
- Ammonium

The clarification behaviour is explained by subdivision in settleable and non-settleable fractions. A model compatible with the ASM No.1 is described by Gernaey et al. (2000). It is based on the approach of Takács et al. (1991), especially regarding the behaviour of particulate and soluble COD.
More advanced descriptions are needed to model the secondary clarification process. Ekama et al. (1997) distinguish between 0- to 3-dimensional models. Regarding the 0-D models one assumes two different streams occurring in the clarifier i.e. a concentrated and a (nearly) solid-free stream. The height of the sludge bed can be predicted by assuming a linear concentration profile based on the

advanced 0-D model by Holzer and Krebs (1998). The 1-D model presented in Takács et al. (1991) is the best known and applied representation that describes particle separation and mass accumulation along the vertical axis. Higher dimensional models (describing the flows and turbulence inside the settling tank in detail in two-dimensional and three-dimensional way) certainly need more variables and equations that are leading to longer computing times. They are applied to understand the interactions between flow and settling behaviour and are being used for designing of the inlet, outlet and return sludge systems (Krebs et al. (2000)). On the other hand 0- and 1-D models allow to describe the transport of sludge between the bioreactor and the clarifier during high flows and are generally utilized for simulation of a complete treatment plant.

3 APPLICATION OF INTEGRATED MODELS TO IMPROVE THE URBAN WASTEWATER SYSTEM

Today one is able to solve complex wastewater management problems by integrated model simulation with different available software packages. By replicating reality and implementing real-time control strategies to optimise the wastewater system processes one is capable to provide a reasonable decision support to optimise and improve the urban wastewater system and eventually the quality of the receiving water body.

A general approach of problem solution, i.e. system enhancement, can be pointed out. First, the existing problem has to be clearly identified with aim of defining the precise aim that should be pursued. Afterwards, by dint of different hydrological, hydrodynamic and water quality models a reasonably realistic description of the given wastewater system can be carried out by implementing it in a modelling and simulation software (WEST® for example). While using the software, simulations can be conducted whose results constitute the basis for development of reasonable strategies, e.g. real-time control (RTC) to improve the receiving water quality.

Schütze et al. (2002) defined real-time control as follows: "An urban water system is controlled in real time if process variables are monitored in the system and continuously used to operate actuators during the process." Thereby parts of a system are manipulated by using control strategies with predefined set points and rules. The measured state of a system (e.g. water level, flows, etc.) and also inputs (e.g. measured rain data) may be used within the applied control algorithm (Butler and Schütze (2005)). Depending on what is manipulated and which variables influence the manipulation, one can distinguish between different approaches of RTC, like volume-based RTC, pollution-based RTC or immission-based RTC (Vanrolleghem et al. (2005)). To test the robustness of the control strategy different scenarios (e.g. water quality changes, changes in treatment processes, addition of retention basins) are simulated afterwards to get an idea of how the real-time controlled system will react when changes in the system are taking place. Thereby a reasonable assessment can be carried out by using multi-criteria evaluation (Muschalla (2006)). In this manner, primary criteria are typically given by online measurements of water quality (usually the component that has to be improved e.g. the ammonia concentration in the receiving water body). The secondary criteria relate to all other variables one wants to investigate (these can also be investment costs) but which do not have direct impact on the controller (Vanrolleghem et al. (2005)). The less the examined criteria are influenced by the simulated scenarios, the more robust is the implemented RTC strategy. Therefore, a comparison of different strategies is essential to decide whether a certain control strategy of the integrated urban wastewater system is really suitable with respect to the particular river ecosystem on the one hand and the economic efficiency on the other hand.

In the following, different case studies will be presented to point out how integrated modelling was used to improve urban wastewater systems and how the evaluation of the improving strategy was realised.

3.1 The Odenthal case study; Erbe et al. (2002)

Within the Odenthal case study Erbe et al. (2002) describe an interesting example for improving the urban wastewater system by using the integrated approach and real-time control. To improve the water quality of the river Dhuenn an integrated system is considered, i.e. not only the wastewater treatment plant, which did not meet the German effluent standards for nitrogen concentration, was taken into account. For a reasonable amelioration of the receiving water quality, the interaction between catchment, sewer, WWTP and river was investigated.

In their study the transition from dry weather to wet weather loading is to be smoothened, while the CSO discharges to the river Dhuenn are not to be increased. The different parts of the considered system were modelled on the one hand by using water balance equations to describe the catchment (Risholt (2000)) and on the other hand by implementation of the ASM No.1 (Henze et al. (1987)) to represent the biological processes of the wastewater treatment plant. Furthermore, for secondary clarification a one-dimensional model was used (Otterpohl (1995)). The river quality was simulated with a submodel of the RWQM1 (Reichert et al. (2001)) to describe conversion processes for dissolved nitrogen and phosphorus compounds based on the assumption of constant concentration of biomass and algae.

The first objective was to simulate the various substance flows in the system to gain an overview of the pollution loads from the three main subcatchments and to further analyse the sensitivity of the river to CSO (combined sewer overflow) events. Afterwards, control strategies were tested to improve the system. Regarding the concentration of free ammonia and nitrite, the best results were found for an operating strategy in which the combined waste load to the WWTP is increased based on online estimates of the nitrification capacity. This was combined with controlling the operation of the wastewater retention in the sewer system (Erbe et al. (2002)). The dose-effect matrix for free ammonia (where frequency is plotted versus time of duration) was applied to evaluate the results in terms of river water quality. The concept of the matrix is to evaluate concentration values against thresholds that depend on duration time and frequency of occurrence (Hoppe et al. (2004)).

3.2 THE HEUSENSTAMM CASE STUDY; MUSCHALLA (2006)

Another example of the amelioration of the receiving water quality by using the integrated approach is given by Muschalla (2006). In his thesis he developed a simulation model for the optimisation of the urban wastewater system of the town Heusestamm (Germany). The quality of the considered water body ranges from moderately polluted (water quality category[1] II) to heavily polluted (water quality category[1] III). To assess the improvement of the system, it is examined by a multi-criteria evaluation focussing on three main criteria: hydraulic discharge, discharge load (ammonia concentration, oxygen deficit) and costs. Different strategies were evaluated to reduce capacity overloads of the combined sewer system. The establishment of local infiltration areas, i.e. a reduction of the impervious surface, was simulated to get an idea how the system would be influenced when the rainwater run-off is reduced. Furthermore, the effect of a throttle flow and additional retention basins scenarios in the sewer was also analysed (Muschalla (2006)). The results with respect to reduction of nutrient loads in the river were compared to the associated investment costs to deliver a reasonable recommendation for ameliorating the river quality.

3.3 THE TIELT CASE STUDY; MEIRLAEN (2002)

Any optimisation study in environmental management is multi-criteria in nature. One has to take more than one objective into account when evaluating control strategies reasonably. Thereby, on-line measurements of the water quality are supplemented by secondary objectives one is interested in. The latter has no direct influence on the controller (Vanrolleghem et al. (2005)). Meirlaen (2002) developed different immission-based control strategies for integrated wastewater systems. Regarding the Tielt case study he focused on the maximum ammonia concentration in the river, while the minimum oxygen concentration was considered as the secondary objective. To model the

[1] The German saprobic assessment of surface water is divided into 4 main-classes; the sectoral water quality category represents the load of readily biodegradable organic matter

considered Tielt catchment (Belgium) a simultaneously simulating model was applied in a software package (WEST®) that is able to replicate the three subsystems in parallel (thus problems of file or data transfer between the subsystems could be avoided). The investigated wastewater treatment plant was not able to ensure a good water quality by only using static operation. Hence, two control strategies were developed to improve the water quality without rebuilding the sewer system. The first control strategy uses measurements in the river to act on the WWTP operation. The second strategy extends the first one, by also acting on the sewer system (Meirlaen (2002)). The reference control strategy is acting on a storm tank at the WWTP. If the designed capacity for biological treatment is reached the inflow is redirected to the storm tank. Once this tank is filled the water is bypassed to an overflow station to spill in the river. Meirlaen describes the first dynamic control strategy as follows. To eliminate peak ammonia concentrations in the river an ammonia sensor is located downstream of the WWTP nearby the overflow station. The controller avoids overloading of the treatment plant during high flow events as long as the ammonia content in the river remains below a critical value (a given set point), i.e. it allows bypassing to the overflow stations in this case. When the critical ammonia concentration in the river is exceeded, the bypass will be reduced and the flow to the biological treatment is increased. Thus, the inflow to the WWTP is proportional to the difference of the current ammonia concentration in the river and a predefined critical value (set point). The second control strategy performs an optimisation of the retention capacity in the sewer system. Thereby, a pump is installed which sends more water downstream as long as the storage tank is not completely filled and the WWTP is not overloaded. It aims to better utilise the hydraulic capacity of the pipe and therewith the retention capacity of the tank.

An evaluation with the integrated model was carried out to test the robustness of the system. This was conducted on the one hand with respect to water quality changes and on the other hand with respect to changes in treatment performance. Regarding the latter three different changes to the system properties were simulated. First, the growth rate of the nitrifying (autotrophic) biomass (μ_A) was temporarily reduced by 10 %. Next, the impact on the system of a 10 % increase of the aeration capacity (K_La) was checked finally a 10% increase of the sewage nitrogen concentration (TKN) was examined. In each of the scenarios the influences on oxygen and ammonia concentration, chemical oxygen demand (COD) as well as the change of flow rate were investigated. Furthermore, the robustness towards two water quality criteria was also evaluated: the maximum ammonium concentration and the time the ammonium was above a certain limit. It was shown that the control strategies had about the same robustness as the reference strategy and could therefore be accepted for implementation (Meirlaen (2002)).

3.4 The Bleesbruck case study; Solvi (2007)

In the PhD study of Solvi (2007), whose work is the basis of this thesis, the evaluation was carried out both from an immission and on emission point of view (Solvi et al. (2005)). An activated sludge wastewater treatment plant at the considered Bleesbruck catchment (Luxembourg) has been previously modernised with tools for real-time monitoring and control of the treatment processes (Schosseler et al. (2000), Schosseler et al. (2003)). Thus, a higher treatment efficiency was achieved. Solvi (2007) conducted an immission-based optimisation regarding the operation of the Bleesbruck sewer network and the WWTP i.e. the influences of the integrated urban wastewater system on the receiving water quality were evaluated. The system model was applied to simulate different strategies which were compared to one another by using long-term simulations over one or two years. Thereby, a large number of scenarios were chosen to be simulated and tested in WEST®. The strategies range from source control measures to construction of infrastructures within the sewer or WWTP, operation measures in one of the subsystems or special measures like shading and reaeration in the river. The simulation results from the second biology were used for evaluating the overall emission performance of the WWTP and as input data to the river model (RWQM1, Reichert et al. (2001)) simulations to evaluate the immission criteria. The scenarios are evaluated on a long and short term basis to describe the impact of “normal” and extreme CSO events on the resulting WWTP performance and river quality. Dissolved oxygen (DO), ammonia (NH_3), phosphate (PO_4^{3-}) and chemical oxygen demand (COD) were chosen as evaluation parameter to assess the particular integrated simulation scenario and to find out which strategy is best. After analysing the results in an ecological and economical context, a decision support for suitable operation and control of the integrated urban wastewater system was delivered to improve the river water quality.

An overview of the main facts of the different case studies is shown in Table 1. It can be recognized that the general approach to improve the receiving water quality focuses on two main strategies: Reduction of run-off and retention of peak flows.

TABLE 1 COMPARISON OF STRATEGIES AND THEIR EVALUATION CRITERIA WHICH WERE DEVELOPED AND INVESTIGATED WITHIN DIFFERENT CASE STUDIES TO IMPROVE THE RIVER WATER QUALITY

Case study	RTC	Strategies	Evaluation criteria
Odenthal	✓	Control of the combined waste load depending on the WWTP nitrification capacity	NH_4^+, NO_2^-
Heusenstamm		Reduction of run-off, retention of wastewater, throttle flow	Hydraulic discharge, NH_4^+, DO
Tielt	✓	Control of the combined sewer load depending on the hydraulic capacity of the WWTP and NH_4^+ concentration in the river	NH_4^+, DO, COD
Bleesbruck		Reduction of run-off, buffer of domestic and industrial wastewater, retention of combined wastewater, optimizing of treatment operations, river shading, artificial reaeration of the river water	NH_4^+, PO_4^{3-}, COD, DO

4 INVESTIGATION OF READILY AND SLOWLY BIODEGRADABLE SUBSTRATE WITH RESPECT TO DIURNAL FLUCTUATIONS IN THE SEWER SYSTEM

4.1 INTRODUCTION

The increasingly recommended integrated study of urban wastewater systems is realized by the use of different dedicated models as already described in a previous part of this thesis. With respect to wastewater quantity and quality, an overview was given of the different kinds of models which are presently used in current research and field application to optimise and improve the integrated system.

One of the most important factors by which a sewer model can be evaluated is how its simulation is capable of predicting the biodegradable substrate loads from the sewer system as it will determine the performance of the treatment plant (e.g. for denitrification).

A possible wastewater quality modelling approach in integrated sewer systems considers sewer components as particulate and soluble material (for example organic material is analysed as soluble and total chemical oxygen demand). Hourly variations, that result from domestic water use and industrial wastewater discharge as well as the size of the catchment area, are used to predict the fluctuation of the sewer compounds during the course of a day. Often, only physical processes like erosion and sedimentation are taken into account to describe influences on concentration fluctuations.
However, regarding the biodegradable substrate fractions in wastewater it can be assumed that these components are also affect by biochemical processes such as hydrolysis and denitrification.

The former describes the bacterial degradation of organic material which is present in the form of particles, colloids or high-molecular compounds. Yet, these materials are not directly available for bacteria. Therefore, the microorganisms release enzymes that degrade such organic matter into water soluble constituents which are directly convertible by biomass. Consequently, hydrolysis contributes to a change of the readily and slowly biodegradable substrate ratio in the sewer system.

Through possible oxygen limitation in sewer systems denitrification processes can also occur which do not only influence the nitrate concentration in wastewater but also the organic matter. Heterotrophic biomass oxidises organic matter whereby nitrate instead of oxygen is reduced (see the qualitative equation (6) for formaldehyde (Gujer (2007)). Consequently, in larger sewer systems the denitrification process can have a notable impact on nitrate and biodegradable substrate concentration.

$$5CH_2O + 4NO_3^- + 4H^+ \rightarrow 2N_2 + 5CO_2 + 7H_2O \tag{6}$$

Regarding wastewater quality at the WWTP inlet, in-sewer degradation could affect planning costs of WWTP design and has notable relevance for WWTP operation with respect to COD, N/COD ratio as well as P/COD ratio (Ahnert et al. (2005)).
Thus, the biodegradable substrate fractions can fluctuate during a day and the mentioned processes may cause variations of the static load pattern that is generally used to model the time evolution of the sewer components during the day.

A typical COD variation during a day is shown in Figure 4 (Solvi (2007)). The diagram shows a highly significant COD fluctuation for the distribution is an example of a relatively small catchment (approx. 34 000 PE) with considerable industrial influence. However, it can easily be recognized that the highest COD concentrations are expected around 6.30 a.m., between 12.00 and 1.00 p.m. and 7.00 o'clock in the afternoon. The lowest COD concentration is present in the sewage at night and in the early morning.

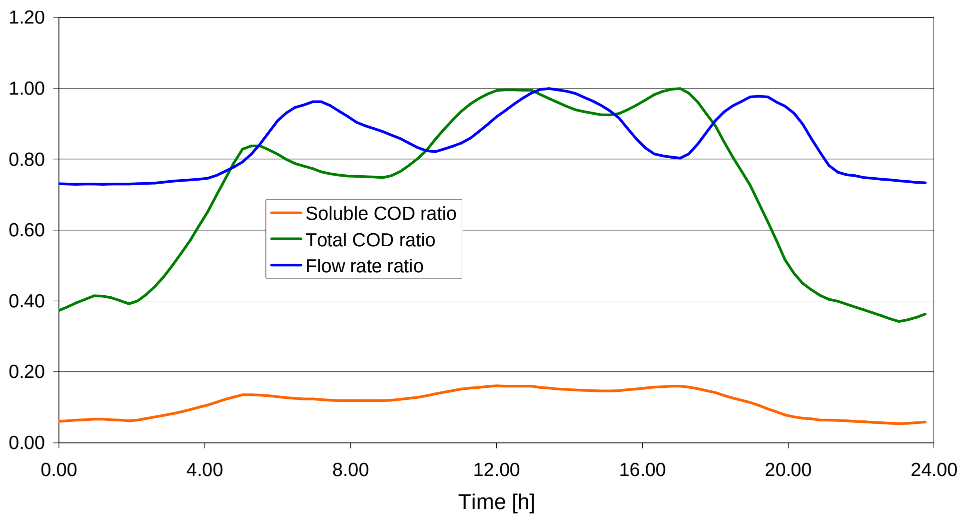


FIGURE 4 DIURNAL FLUCTUATION OF TOTAL AND SOLUBLE COD AS WELL AS FLOW RATE VARIATION OF THE BLEESBRUCK SEWER MODEL (SOLVI (2007)) DURING A DAY WHEN DRY WEATHER PREVAILS (SOL COD / TOT COD = 0,16)

When dry weather is prevailing the soluble COD pattern shows a similar distribution if only physical processes are taken into account to predict the diurnal substrate fluctuation. Regarding the Bleesbruck sewer model, only sedimentation processes are described within storm water tanks and pipes which influence the soluble / total COD ratio. All other biological effects are not implemented in the model.

The context described above was the incentive for the following laboratory analysis to verify diurnal fluctuations of biodegradable substrate fractions in sewer systems which can have a significant impact on the treatment performance especially because the biological phosphorus removal and denitrification process at the WWTP are sensitive to those variations.

The biodegradable substrate is partitioned into two fractions: readily and slowly biodegradable matter. The former is a soluble part of the chemical oxygen demand (COD) and is directly available for microorganisms which transform the substrate into cell growth and energy for cell maintenance (see Figure 5). The latter consists of high molecular weight, colloidal and particulate organic compounds which have to be transformed through processes of external cell hydrolysis (as already described) before they are available for degradation (Henze et al. (1999)). Regarding its concentration in wastewater the physical characteristics of this COD fraction lead to a higher dependency on flow rates in the sewer system. It is assumed that parts of the slowly biodegradable matter are subject to sedimentation and erosion effects. However, further investigations have to be carried out to give a more suitable description of this COD fraction to finally improve WWTP simulations.

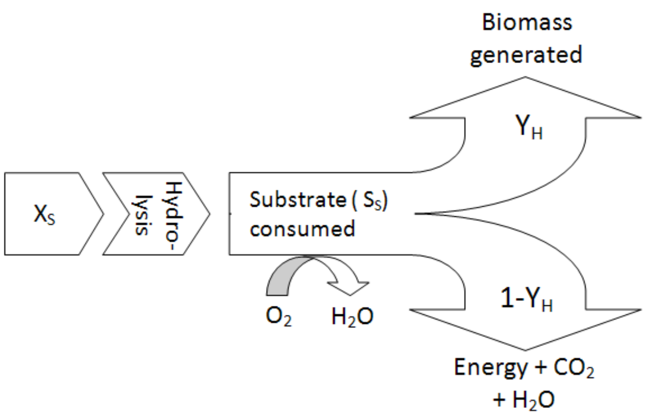


FIGURE 5 SCHEMATIC OF MICROBIAL SUBSTRATE DEGRADATION

With laboratory analysis one is capable of determining the two described fractions of biodegradable COD. Respirometry provides a suitable measurement method and interpretation of the biological oxygen consumption rate under well-defined experimental conditions (Spanjers et al. (1998)). Measuring the oxygen consumption rate allows drawing conclusions about slowly and readily biodegradable substrate concentrations in wastewater.

Thus, an experimental setup was developed to examine the dependency of the slowly and readily biodegradable substrate concentration at different times of the day. With the investigation of different wastewater samples the diurnal variation of slowly and readily biodegradable matter is to be shown.

4.2 The Saint – Nicolas wastewater treatment plant

The investigated wastewater and activated sludge were taken at the municipal Saint-Nicolas wastewater treatment plant nearby Québec City (See Figure 6) which receives wastewater from three subcatchments with a population of 36 000 inhabitants (Charny, Saint-Nicolas and Saint-Rédempteur). The physical and chemical characteristics of Saint-Nicolas wastewater which were analysed in a previous master thesis (Aubry (2003)) are presented in Table 2.

FIGURE 6 CATCHMENT AREA OF THE SAINT-NICOLAS WWTP (HTTP://MAPS.GOOGLE.COM/)

The large standard deviations observed reflect the variability of the raw water characteristics and show that individual characterisation of each sample is necessary.

The plant was built in 1997 to treat 18 760 m³ wastewater per day. The diurnal mean load of biodegradable organic matter (2227 kg BOD_5) is handled by the use of a sequencing batch process. Thereby, three reactors (each with a volume of 3500 m³) are alternately charged.

The SBR process is a type of activated sludge treatment. Compared to a conventional continuous flow process the treatment steps do not follow a treatment line (cascade treatment). Rather the processes are controlled in time all necessary biological treatment processes are not taking place separately in different reactors (anaerobic, aerobic, anoxic phase, secondary clarifier) but sequentially in the same reactor. The mechanical pre-treatment where suspended solids are being settled, occurs at the upstream buffer tanks (two times 1305 m³) which also serve as retention for wastewater.

TABLE 2 SAINT NICOLAS WASTEWATER CHARACTERISTICS ANALYSED IN THE WORK OF AUBRY (2003)

	Average*	Standard deviation	Number of measurements
$NH4^+$-N	21.9	8.6	51
Total COD	282	78	55
Filtered COD	130	38	47
TSS	136	98	43
Alkalinity	191	27	12
pH	7.7	0.3	33

* Expressed in [mg/L] except Alkalinity [mg$CaCO_3$/L] and pH

The installation at the Saint Nicolas wastewater treatment plant consists of three identically equipped tanks with a common inlet, which can be switched among them to allow continuous treatment. The reactors follow different sequential operating modes: While one tank is in settle/decant mode another is aerating and filling. Aeration times can vary between 60 and 90 minutes according to the composition/quantity of the incoming liquor at the Saint Nicolas WWTP. The aeration process encourages the growth of autotrophic aerobic bacteria that transform ammonia nitrogen to nitrate nitrogen.

During the settling stage the sludge formed by the bacteria is allowed to settle to the bottom of the tank. The aerobic bacteria continue to grow until the dissolved oxygen is used up. The ensuing anoxic conditions are more suitable for heterotrophic bacteria (facultative anaerobe) which are able to use

nitrate nitrogen (created in the aeration stage) as a oxygen source and organic substrate as carbon source to grow and to produce energy. This stage is known as the denitrification.
As the bacteria multiply and die, the sludge within the tank increases over time and a waste activated sludge pump removes some of the sludge during the settle stage to a digester for further treatment. The sludge is allowed to settle until clear water is on the top 20%-30% of the tank contents.
(A process chart of the WWTP St-Nicolas is presented in Appendix A)

4.3 Wastewater analyses

Wastewater samples were taken right after the sand removal unit and the same amount of activated sludge was sampled from the sequencing batch reactor (SBR).
All the experimental conditions and respirometer tests described in the following refer to the work performed in the Water Laboratory (labEAU) of Université Laval. During the test period, the wastewater and activated sludge samples were stored at 4°C to avoid significant chemical and especially microbial processes to take place.

4.3.1 Total suspended solids (TSS) / volatile suspended solids (VSS)

TSS describes the amount of filterable solids in a water sample. Volatile suspended solids are those particles lost on ignition (heating to 550 °C). They are analysed since they give a rough approximation of the amount of organic matter present in the suspended fraction of wastewater and activated sludge.

TSS and VSS analysis were carried out by using a vacuum filter apparatus and filter paper with a pore size of <1.5 µm. Well-mixed samples (wastewater: 100mL; activated sludge: 50mL) were filtered through a glass fibre filter. The filters were dried for at least 2 hours at 105°C and weighed to determine the amount of total suspended solids in mg/L of sample. Afterwards, the samples were ignited in a furnace at 550°C for at least two hours. The glass fibre filter papers were cooled down in a desiccator before they were weighed again to calculate the volatile suspended solids fraction of the wastewater and sludge.

4.3.2 Chemical oxygen demand (COD)

The Chemical Oxygen Demand analysis is a means of measuring the pollution strength of wastewater. By using this method, most oxidisable organic compounds in the wastewater and sludge samples may be measured.

The COD test has a significant advantage over the biochemical oxygen demand analysis because of the short time (several hours) that is required to perform the analysis compared to the standard BOD (Biochemical Oxygen Demand) test where five days are needed. By principle, COD values are higher than BOD values. This is because biochemical oxygen demand only measures the quantity of organic material that is oxidized. The other part is incorporated into biomass or is not biodegraded. The COD on the other hand represents a more complete oxidation. A reference value for a typical COD concentration in a combined sewer system is 300 mgCOD/L (Bischof (1998)).

After calibrating the spectrophotometer with well-known COD solutions (see diagram in Appendix C) several COD tests with filtered (filter pore size <1.5 µm) and non-filtered samples of wastewater and activated sludge were examined for total and soluble chemical oxygen demand. In order not to exceed the range of 0 – 1500 mgCOD/L of the used COD vials, the non-filtered sludge samples were diluted ten times with distilled water. All COD tests were carried out according to the standard procedure found in appropriate literature (HACH® method 8000) (HACH® (2003)). After heating the samples in a COD reactor (HACH®) at 150°C for 2 hours and cooling them down, the absorbance at a wavelength of 600nm were measured in the spectrophotometer (Ultraspec III, Pharmacia LKB).

4.3.3 Ammonia nitrogen (NH_4^+-N)

To investigate the influence on oxygen consumption due to nitrification during the respirometry tests, the soluble ammonia nitrogen concentration was measured according to the HACH® method 8038 (HACH® (2004)).

For this, filtered (pore size: 0.45 µm) wastewater and activated sludge samples were diluted ten times with distilled water and conveyed in a 25 mL vial. Afterwards, 3 drops of mineral stabilizer were added to each sample. The vials were well-mixed and 3 drops of polyvinyl alcohol dispersing agent were added and blended afterwards. Then 1 mL of Nessler reagent was injected. The vials were well-mixed again and after one minute the sample was placed in the spectrophotometer (Ultraspec III, Pharmacia LKB) and analysed at a wavelength of 420 nm.

The spectrophotometer was calibrated before these analyses by using well-known NH_4^+-N concentration solutions (see Appendix C).

4.3.4 RESPIROMETRY

4.3.4.1 RESPIROMETER - SETUP

Basically the respirometer BIOS-R (developed by INSA Toulouse, France) consists of 5 units (see Figure 7). The reactor module (volume: approx. 2.6 litre) comprises the bioreactor, the pumps that regulate the pH value and the pH probe (H8481 HD, Schott).

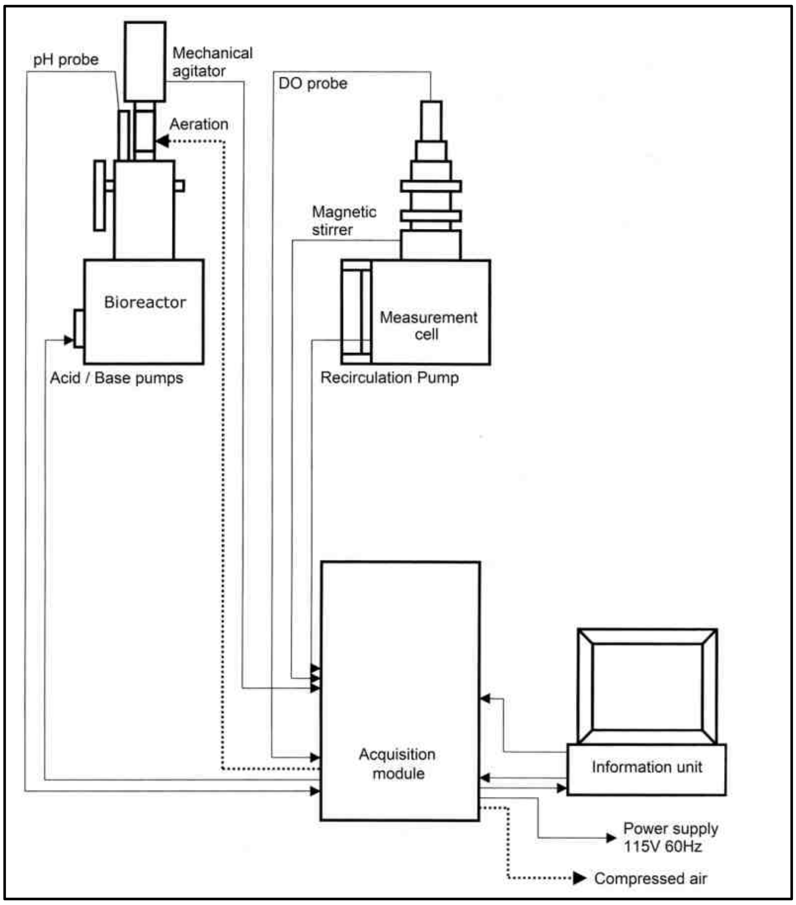


FIGURE 7 RESPIROMETER SETUP IN PRINCIPLE (WITHOUTTHERMOSTATIC BATH)

The oxygen meter (Quadroline Oxi 296, WTW), the pH-meter (Quadroline pH 296, WTW) and the closed loop for oxygen regulation are located in the acquisition module. At certain instants, part of the investigated mixed liquor is conveyed from the bioreactor into the measurement cell (Volume approx. 400 mL) that includes the required pump and a DO probe (TriOxmatic® 701, WTW).

The thermostatic bath (NESLAB RTE 7, Thermo Electron Corporation) controls the temperature of the bioreactor and measurement cell which are double-walled with a recycling water flow. A computer is linked to the acquisition module and continuously collects data and manages all instructions.

FIGURE 8 RESPIROMETER SETUP IN THE WATER LABORATORY (LABEAU) OF UNIVERSITÉ LAVAL

4.3.4.2 RESPIROMETER – OPERATION MODE

In principle, the respirometer can be considered as a measuring device for fast BOD (biological oxygen demand) tests. That is, by using a DO probe it is possible to measure the decrease of oxygen concentration due to substrate degradation in a non-aerated, well mixed sample over a certain time interval.

A certain volume of sludge / wastewater mixture is continuously aerated with an oxygen flow of 60 L/h in the bioreactor. Furthermore, a mechanical agitator (RW 16 basic, IKA® Werke) ensures with a rotation of 360 rd/h that the reactor content is well mixed. Two tubes provide the reactor with sodium hydroxide (NaOH; 0.1 mol/L) and sulphuric acid (H_2SO_4; 0.05 mol/L) to keep a relatively constant pH-value during the test. The acquisition module linked with the computer controls the base and acid supply.

The pH-range of the following tests was chosen between 7.0 and 7.5 since the pH-value of the wastewater at the treatment plant is in the same range.

The thermostatic bath keeps the temperature at 25°C in the bioreactor as well as in the measurement cell to ensure good microbial activity.

After a certain interval a recirculation in the measurement module is conducted. That means 400 mL of mixed liquor is leaving the cell and is led back into the bioreactor and simultaneously a "new" 400 mL sample is pumped from the bioreactor into the measurement cell. Since there is no oxygen provided in the cell the decreasing oxygen concentration can be examined while the sample is continuously mixed at 300 rd/min by using a magnetic stirrer device (MR 3000, Heidolph Instruments).

At a certain frequency fixed by the user the computer acquires oxygen concentration values and during each recirculation the device calculates the oxygen uptake rate (OUR) from the negative slope of the measured oxygen concentration (see Figure 9).

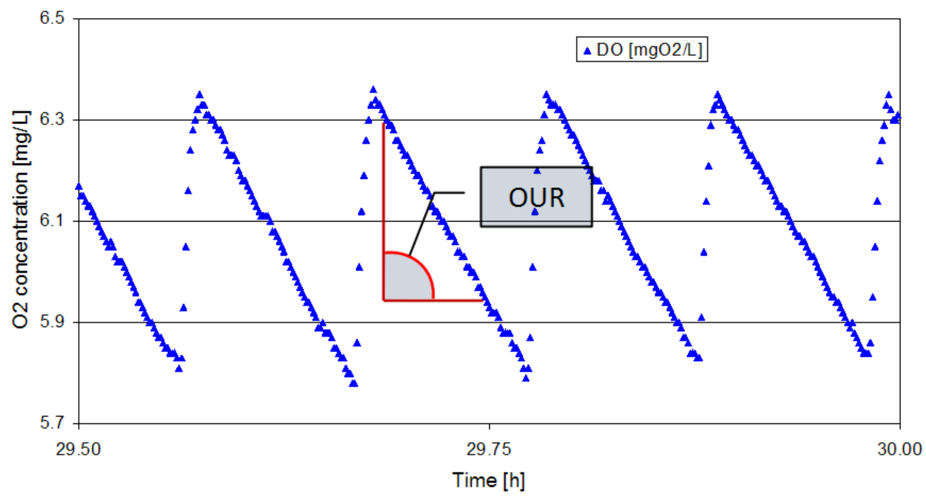


FIGURE 9 EXTRACT OF THE DISSOLVED OXYGEN CONCENTRATION IN THE MEASUREMENT CELL DURING A RESPIROMETER TEST.

Before the beginning of each test the user has to verify or to enter the recirculation frequency and the interval of DO value acquisition. With respect to the latter, an interval of 5 seconds was chosen to assure a reasonable calculation of the decreasing oxygen slope still allowing a sufficient long collection time limited by the fixed database size.

For the recirculation frequency two criteria have to be specified: First, the minimum dissolved oxygen concentration in the measurement cell, which was defined as 2 mg/L since limited oxygen concentrations are not wanted if heterotrophic aerobic degradation is to be investigated. The maximal residence time in the measurement cell is the second criterion, whereby for the conducted tests was set to five minutes.

Depending on which critical value is reached first, a recirculation between the two modules is performed.

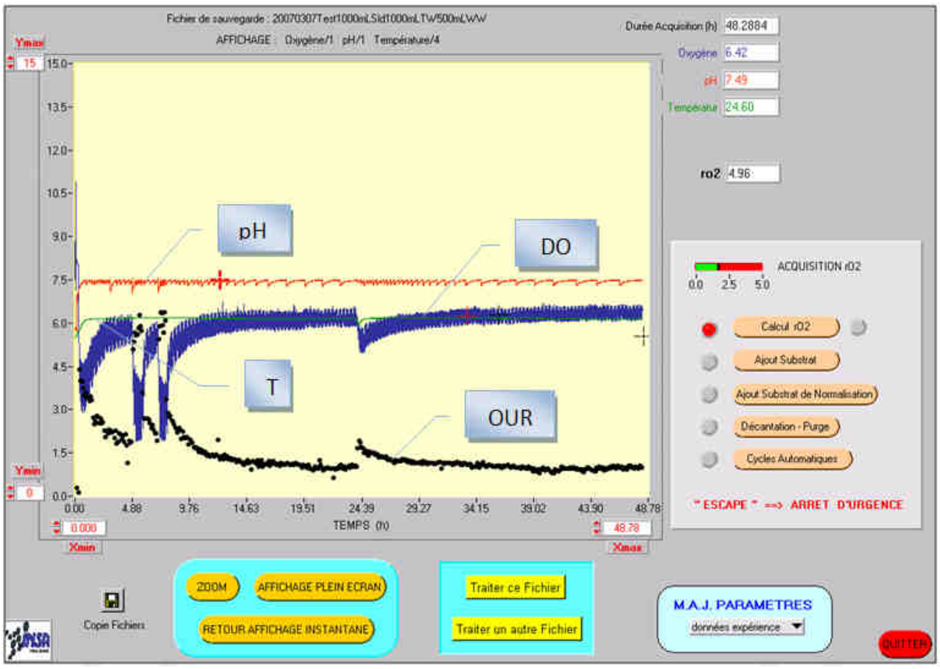


FIGURE 10 USER INTERFACE OF THE RESPIROMETER (SHOWING FROM TOP TO BOTTOM: PH-VALUE, DISSOLVED OXYGEN CONCENTRATION, TEMPERATURE AND OXYGEN UPTAKE RATE)

4.3.4.3 RESPIROMETRIC PRE–TESTS

To examine the respirometer performance as function of different mixtures of wastewater and sludge some so-called pre–tests were conducted. The purpose was to find a reasonable ratio of wastewater and activated sludge since the biodegradable substrate concentration in the wastewater and the microbial activity of the sludge were unknown.

Wastewater and activated sludge were sampled in 10-litre vessels at 10 o'clock on Monday (2007/03/05) at the Saint Nicolas plant. Regarding the weather conditions at the time of sampling were dry. A condition of melting snow can be excluded since the air temperature was around minus five degrees.

Each time the same procedure was carried out. First, a particular amount of activated sludge was put into the respirometer vessel, aerated (60L/h) and mixed until the endogenous respiration was reached after several hours. This condition can be defined as the oxygen consumption rate in the absence of substrate from external sources (Spanjers et al. (1998)). The endogenous respiration is practically independent of substrate concentration and consequently indicative of the active biomass (Spanjers et al. (1998)). The condition occurs when the oxygen uptake rate in the respirometer is relatively stable. When this endogenous state is reached, a suitable amount (18mg per litre sludge) of ATU (Allylthiorea) was added to inhibit nitrification during the specific investigation of heterotrophic substrate degradation.

After approx. 30 minutes the substrate samples were added into the respirometer vessel. On the one hand this was conducted with a reasonable amount (see below) of volatile fatty acids (VFA) (which present only readily biodegradable substrate) to evaluate whether the sludge possesses a certain activity as well as to determine the maximum respiration rate (explained in the next paragraph). On the other hand a certain amount of wastewater (that includes readily as well as slowly biodegradable matter) was used to finally measure the microbial oxygen uptake rate against time when substrate is degraded for microbial growth until all substrate is consumed and the endogenous respiration is reached again.

The amount of VFA to be added was calculated as follows:

- Presuming a high microbial activity: 50 $mgO_2/(L*h)$ will be consumed when readily substrate is added. The process is designed to last for 2 hours so as to have a nice respirogram from the respirometer used here (see Figure 11).

  $\rightarrow 50\frac{mgO_2}{L*h}*2h = 100 mgO_2/L$

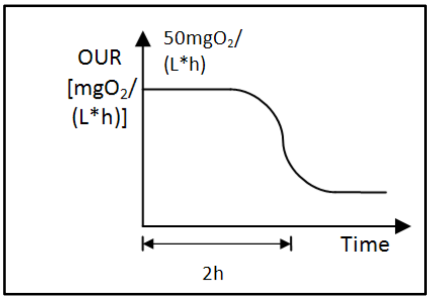


FIGURE 11 SCHEMATIC OF COD FLOW

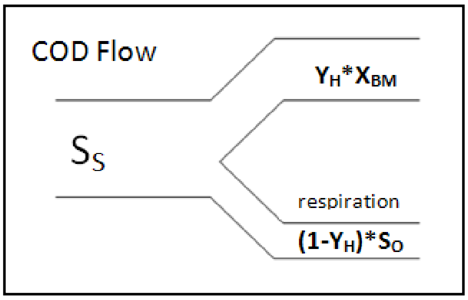


FIGURE 12 ASSUMPTION OF THE MICROBIAL ACTIVITY IN THE SLUDGE

- According to Henze et al. (1987) one assumes that only one third (Y_H = 2/3) of a certain substrate amount is oxidized into energy (the part which is measured during the respirometer test) while the other fraction is utilised for biomass generation (see Figure 12).
  → $100\,mgO_2 / L * 3 = 300\,mgCOD$ is needed for 1 litre sludge
- The used VFA solution contained 108 gCOD/L
  → 3 mL of the VFA solution has to be added to one litre sludge (324mgCOD/L sludge) in order to gain the assumed oxygen consumption of 50 mgO$_2$/(L*h) for 2 hours

4.3.4.4 RESPIROMETRIC TESTS

Samples for the respirometric tests were taken at three different times on Thursday, 29th of March at the Saint Nicolas wastewater treatment. To measure a significant fluctuation of readily and slowly biodegradable substrate the sampling was carried out in the morning, noon and afternoon. The activated sludge sample was taken in the morning and used for the whole test period. It has to be mentioned that the sludge was removed 10 minutes into the aeration phase of the SBR after the reactor filling. This could cause a high ammonia and slowly biodegradable substrate concentration and should be considered during the preparation of the respirometer tests.

As shown in Table 3 snow melting was prevailing during the sampling day. That may have led to a notable dilution of the wastewater, leading to a lower biodegradable substrate concentration.

TABLE 3 WASTEWATER AND ACTIVATED SLUDGE SAMPLING DETAILS

	Time	Volume of sampling vessel	Sampling location	Weather condition
WW Sample 1	07.45	2 * 5L	After sand removal unit	Sunny, -5°C
WW Sample 2	12.00	2 * 5L	After sand removal unit	Sunny, 3°C (snowmelt)
WW Sample 3	15.30	10L	After sand removal unit	Sunny, 6°C (snowmelt)
Activated sludge sample	07.45	2 * 10L	Sequencing Batch Reactor	

Each wastewater sample was respirometrically analysed twice and stored at four degrees in a climate chamber as it was already done during the pre-tests. Besides the investigation of slowly and readily biodegradable substrate, the ammonia nitrogen concentration, total and soluble COD as well as total suspended solids of the wastewater were also measured (according to the procedures described above).

Before the substrate was filled into the bioreactor of the respirometer, the activated sludge was mixed and aerated during the night to reach the endogenous state. Every second test, a used (from a previous test) activated sludge sample was utilized to reduce that time to approx. 6 hours since the already used samples contained much less biodegradable organic matter than fresh samples from the SBR. In doing so, the mixed liquor from the bioreactor was filled into a vessel where the particulates could settle for 30 minutes. Afterwards, the supernatant was decanted and the remaining 1L sludge was used for the next respirometric test. Furthermore, 18 mg of ATU (Allylthiorea) per litre sludge was injected 30 minutes before the oxygen uptake rate was measured and substrate was added.

4.4 Results and Discussion

In the following three paragraphs the results of the TSS/VSS -, COD - and NH_4^+ analysis are presented. These components were analysed in the course of the respirometric pre–tests to evaluate the wastewater and sludge condition. Three samples of wastewater and activated sludge were analysed. The mean value of the three measurements is given after each paragraph.

4.4.1 TOTAL SUSPENDED SOLIDS (TSS) AND VOLATILE SUSPENDED SOLIDS (VSS)

The wastewater and activated sludge samples were investigated by using a standard procedure which has already been described in paragraph 4.3.1.

Table 4 shows that after the sand removal unit at the Saint-Nicolas WWTP the largest fraction of suspended solids consists of organic matter (VSS). The VSS/TSS ratio ranges between 81% in the activated sludge to 86% in the wastewater.

TABLE 4 TSS AND VSS FRACTION OF THE INVESTIGATED WASTEWATER AND SLUDGE

	TSS [g/L]	VSS [g/L]	VSS/TSS
Wastewater	228	196	0,86
Activated sludge	1975	1601	0,81

4.4.2 CHEMICAL OXYGEN DEMAND (COD)

The activated sludge exhibits a high COD value due to its high concentration of microorganism which are responsible for the organic substrate degradation taking place in the treatment process (Table 5). The wastewater possesses a relatively low (23%) fraction of soluble COD which confirms the assumption that the investigated wastewater has a high ratio of slowly biodegradable substrate compared to the readily biodegradable matter. Thus, one can expect a considerable effect on the following respirometer tests.

TABLE 5 COD RESULTS OF THE INVESTIGATED WASTEWATER AND SLUDGE SAMPLES

	Total COD [mg/L]	Soluble COD [mg/L]	Soluble COD / Total COD
Wastewater	581	131	0,23
Avtivated sludge	2513	40	0,02

4.4.3 AMMONIA NITROGEN (NH_4^+-N)

As expected, a higher (approx. 2.5 times) ammonia concentration was found in the wastewater (32.27 mgNH_4^+-N/L) than in the activated sludge samples (13.70 mgNH_4^+-N/L).
Thus, a certain amount of ammonia was present and this has to be considered during the respirometry tests where an appropriate amount of ATU (Allylthiorea) has to be added to inhibit nitrifying bacteria that would otherwise affect the oxygen consumption in the respirometer.

4.4.4 RESPIROMETRY

4.4.4.1 RESPIROMETRIC PRE-TESTS

Several respirometer tests were conducted to find a reasonable ratio of activated sludge and wastewater. In this respect, the volumes are dependent on the microbial activity of the sludge and the concentration of biodegradable substrate in the wastewater.

Figure 13 shows an example of the respirometric pre-tests. The test was carried out by using a diluted sludge sample composed of one litre activated sludge and one litre tap water. The dilution was applied due to the finding in a previous test that the sludge activity was really high. This led to high OUR values and therefore to too short respirometric cycles with insufficient DO data to calculate the slope, i.e. the OUR.
Furthermore, it has to be mentioned that the minimum DO concentration of 2 mg/L in the measurement cell and not the maximum time interval of five minutes was consistently the initiator for the recirculation during the respirometric test. Measurements of volatile suspended solids as well as COD concentrations confirmed a relatively high sludge concentration.

After reaching the endogenous respiration, illustrated by the stable dissolved oxygen oscillation, 1.5 mL (as calculated above) volatile fatty acids were added. A significant decrease in oxygen concentration can be observed in the diluted sludge phase. The microorganisms consume oxygen to oxidize the volatile fatty acids. This process takes a relatively short time until the oxygen oscillation reaches again the concentration of the endogenous respiration.

It has to be pointed out that the high biomass activity of the activated sludge nearly matches the previously calculated oxygen consumption rate for the VFA degradation. Therefore, within two hours the added nutrient solution was consumed.

Regarding the temperature and pH-values, it can be stated that both variables were relatively stable during the respirometric tests, except for the expected temperature drop when 0.5 litres of wastewater were added.

On the basis of the relatively small drop in oxygen concentration and the subsequently long period of increasing oxygen content (until reaching the endogenous respiration again), it can be observed clearly that the added wastewater contains a relatively low concentration of biodegradable substrate. Furthermore, the absence of a peak in the oxygen consumption rate indicates that the ratio of readily (S_S) to slowly (X_S) biodegradable substrate is low.

In any case, this assumption will be further analysed when examining the oxygen uptake rate after wastewater addition in the next paragraph.

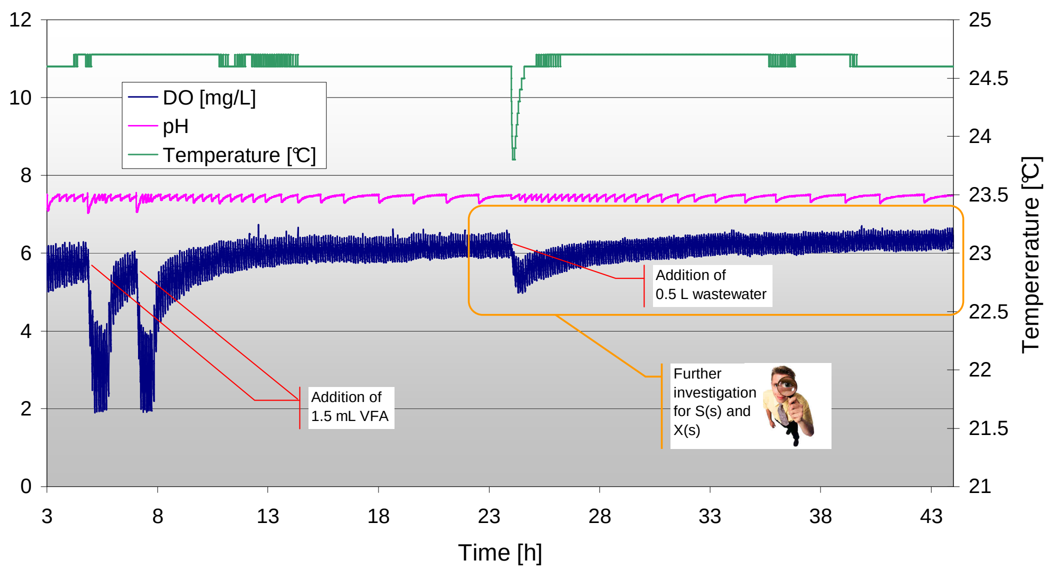


FIGURE 13 DATA SET OF A RESPIROMETRY TEST WITH 1L ACTIVATED SLUDGE DILUTED WITH 1L TAP WATER AND AN ADDITION OF 2 TIMES 1.5 ML VFA AND 0.5 L WASTEWATER

Before analysing and interpreting the oxygen uptake rate (OUR) and the determination of the readily and slowly substrate fractions, the different types of OURs have to be explained with some more detail.

The endogenous respiration rate was already described above. However, one also refers to the actual OUR (a function of the current biodegradable substrate concentration and dependent of the loss due to degradation) and the maximum respiration rate (Spanjers et al. (1998)).

Under realistic conditions of an activated sludge plant treating sewage the latter describes the result of the oxidation of multiple substrates by different kind of bacteria. This leads to the conclusion that the true maximum respiration rate is only reached if all different substrates are present in excess. In activated sludge plants this is very unlikely to occur. Nevertheless, in well-designed respirometric tests the maximum respiration rate can be observed (Spanjers et al. (1998)).

In the experimental setup the maximum respiration rate for heterotrophic bacteria can be examined right after the volatile fatty acids addition.

Thus, the maximal oxygen consumption rate as a result of wastewater addition (see Figure 14) must be considered as the relative maximum that can be achieved with the addition of this particular substrate as analysed below.

The time evolution of the OUR (Figure 14) can be divided in three parts. The constant part of the endogenous respiration is reached when all biodegradable substrate is consumed and oxygen is only used to maintain the biomass.

This condition can be found when the linear decreasing part of the OUR is reaching an intersection point with the endogenous respiration which is observed at 12.15 $mgO_2/(L*h)$ (in the illustrated example (Figure 14) approx 15 hours after wastewater addition).

Furthermore, the beginning of the OUR curve rather resembles a exponential decrease. This part represents the oxygen uptake rate when, in addition to slowly biodegradable organic matter, readily biodegradable substrate is oxidized too. This degradation is much faster than the linear decrease corresponding to the slowly biodegradable substrate degradation. After the readily substrate is consumed, oxygen is only used to degrade the particulate and colloidal organic substrate that is being hydrolysed.

By separately integrating the two decreasing OUR parts (illustrated with green and orange colour in Figure 14) and considering that only a part (1-Y_H) of the substrate is oxidized for energy and can be detected with the respirometer, one is finally able to determine X_S (slowly biodegradable substrate) and S_S (readily biodegradable substrate) (see Table 6).

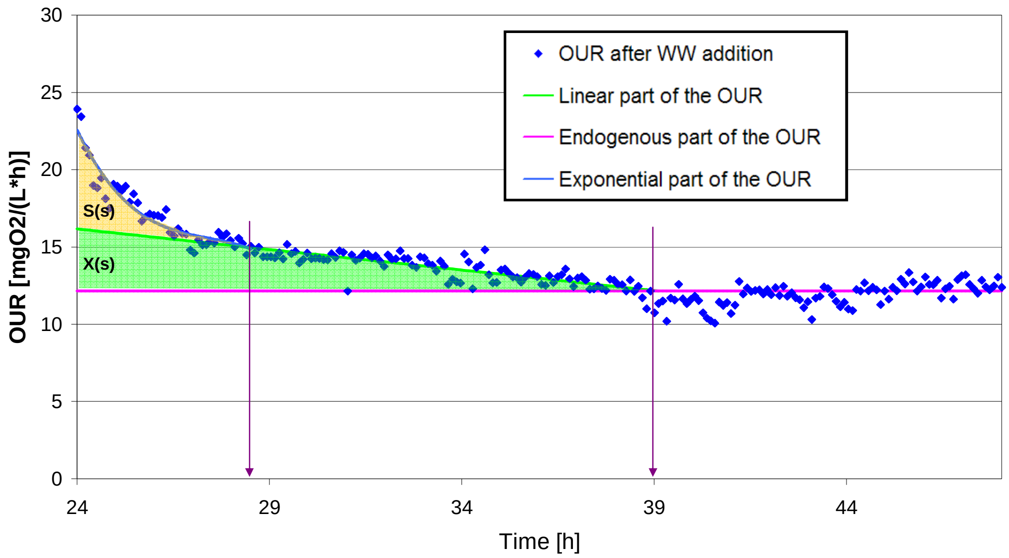


FIGURE 14 CUTOUT OF THE OUR FROM THE TIME OF WASTEWATER ADDITION UNTIL REACHING THE STABLE OUR (VOLUMES: 1 L ACTIVATED SLUDGE, 1 L TAP WATER, 0.5 L WASTEWATER)

Endogenous respiration [$mgO_2/(L*h)$]	12.15
S(s) = ∫ OUR Ss dt / (1-Y_H) [mgCOD/L]	31.58
X(s) = ∫ OUR Xs dt / (1-Y_H) [mgCOD/L]	90.88
Ss / Xs	0.35

TABLE 6 RESULTS OF A REPRESENTATIVE RESPIROMETRY TEST (Y_H = 0.67)

As already presumed, the investigated wastewater exhibits a low concentration of readily biodegradable substrate (31.58 mgCOD/L). This result has to be taken into account for the upcoming respirometer tests in which different wastewater samples collected at different times of the day will be analyzed. To obtain a higher OUR when substrate is injected and to reduce the time used to degrade the COD, a ratio of two litres sludge and one litre wastewater is recommended for the "real" respirometric test.

4.4.4.2 RESPIROMETER TESTS

As already indicated before, the samples for the respirometer tests show a quite low concentration of organic matter as well as ammonia concentration compared to the values that were determined during the pre-tests (See Table 7).

TABLE 7 CONDITION OF THE DIFFERENT WASTEWATERS IN COMPARISON TO THE WASTEWATER THAT WAS ANALYSED DURING THE PRE-TESTS.

	Total COD [mg/L]	Soluble COD [mg/L]	NH_4^+-N [mg/L]
WW (07.45)	185	60	13
WW (12.00)	312	121	11
WW (15.30)	234	96	10
WW pre-tests	581	131	32

The dilution effect due to the snowmelt was also confirmed by the first respirometer test where two litre sludge and one litre wastewater was mixed into the respirometer. The determined maximum of the OUR was very low so that it was decided to increase the substrate volume by using one litre of concentrated sludge and two litre of wastewater for all the following tests. By doing so, two litres of a well mixed sludge sample were taken to let the suspended solids settle down. After 30 minutes one litre of the supernatant was decanted and the remaining concentrated sludge was used for the respirometric tests. This method was chosen to increase the substrate concentration in the vessel without losing activated sludge quantity in the limited volume of the respirometer.

TABLE 8 ACTIVATED SLUDGE CONDITIONS DURING THE RESPIROMETRIC TESTS (THE PRESENTED VALUES ARE FROM THREE MEASUREMENTS)

	New / Used sample	Concentration	Total COD [mg/L]	Soluble COD [mg/L]	TSS [mg/L]
Sludge (1) 07.45	New	/	2112	54	1448
Sludge (2) 07.45	used	2 : 1	2987	127	1815
Sludge (1) 12.00	New	2 : 1	4234	103	2782
Sludge (2) 12.00	Used	2 : 1	3015	99	2033
Sludge (1) 15.30	New	2 : 1	4197	91	2462
Sludge (2) 15.30	Used	2 : 1	2727	84	1838
Sludge pre-tests	New	/	2513	40	1975

The activated sludge that was used for the respirometer tests was being concentrated except for the first test and the pre-test. Hence, it can be observed that the total and soluble COD as well as the TSS concentration is significantly higher than the values that were measured during the pre-tests and the first test where the wastewater sampled at 07.45 a.m. was analysed (See Table 8).

Every second test of the same wastewater, the previous (already used) activated sludge from the respirometer was used again to shorten the time that was needed to reach the endogenous condition before the wastewater was added. Thus, It can be observed that the total COD concentration of the second test from each wastewater is approximately one third lower than the measurement before. The same counts for TSS. This leads to the assumption that during the second test preparation (where the sludge sample from the previous test was used again) some particulates were washed out during the decantation of the supernatant. Furthermore it can be assumed that fresh sludge samples contained biodegradable organic matter (since they were sampled just 10 minutes after the SBR filling at the WWTP). This was not the case regarding the used sludge samples where the organic substrate was already degraded in the previous test. In this context it has to be pointed out that the above described measurements were always carried out before the sludge was filled into the bioreactor where it was aerated to reach the endogenous phase. Hence, at this stage of the test performance differences in respect of biodegradable substrate concentration could have been occurred when both manners are compared where on the one hand fresh activated sludge was used and on the other hand sludge from the previous test.

The results presented in Table 9 exhibit a significant variation of the biodegradable substrate fractions and of the total and soluble chemical oxygen demand. Each analysed wastewater sample shows another concentration and composition of its organic material.

The highest COD load was determined in the 12.00 o'clock wastewater. The readily and slowly biodegradable substrate also shows the highest concentrations (S_S = 63 mgCOD/L and X_S = 90 mgCOD/L) for this wastewater (See Table 9). Nevertheless, in the morning (07.45 a.m.) no dilution due to snowmelt could have had an effect on the results since the temperature was below zero. Not only the lowest concentration of biodegradable substrate was analysed in that sample, but also that of the whole organic material (total COD).

TABLE 9 RESULTS OF THE RESPIROMETRIC TESTS AND SPECTROPHOTOMETRIC COD TESTS COMPARED TO REFERENCE VALUES FROM LITERATURE (X_S AND S_S → AVERAGE OF TWO TESTS, COD → ARITHMETIC MEAN RESULTING FROM 6 MEASUREMENTS)

	Respirometric Analysis		Spectrophotometric Analysis	
	X_S [mgCOD/L]	S_S [mgCOD/L]	Total COD [mg/L]	Soluble COD [mg/L]
WW (07.45)	49	29	185	60
WW (12.00)	90	63	312	121
WW (15.30)	62	61	234	96
WW (ASM2*)	80 - 600	30 - 310		

* Typical range for municipal wastewater according to Henze et al. (2000)

With respect to the given reference range for X_S and S_S (Henze et al. (2000)) it can be pointed out that even the highest concentrations of slowly and readily biodegradable substrate at 12.00 p.m. are relatively low compared to a typical municipal wastewater.

In Table 10 the ratio of each biodegradable substrate fraction to the total organic material of the different wastewater samples is shown. The ratio of S_S to the total COD presents different values for each sample. The same can be stated for the ratio of soluble / total oxygen demand. The slowly biodegradable substrate shows a relatively constant ratio to the total chemical oxygen demand.

Furthermore, it can be recognized that the slowly biodegradable fraction is considerable lower compared to the typical range for municipal wastewater (Henze et al. (2000)). On the contrary, the S_S / total COD ratios of the three wastewater samples fit unexceptionally in the presented reference range.

TABLE 10 RATIO OF X_S AND S_S TO THE TOTAL COD IN COMPARISON TO SOLUBLE / TOTAL COD RATIO OF DIFFERENT WASTEWATERS

	X_S / total COD	S_S / total COD	Soluble COD / total COD
WW (07.45)	0.26	0.16	0.32
WW (12.00)	0.29	0.20	0.39
WW (15.30)	0.26	0.26	0.41
WW (ASM2*)	0.30-0.60	0.12-0.30	

* Typical range for municipal wastewater according to Henze et al. (2000)

The low concentration of biodegradable substrate asked for a relatively extensive analysis of the OUR data acquired by the respirometer since the interpretation of the oxygen uptake rate and determination of readily and slowly biodegradable substrate is carried out graphically. The lower the biodegradable substrate concentration exists in the wastewater the lower the linear and exponential part (See Figure 14) of the OUR become and the more sensitive the influence is on the X_S / S_S partition made by the trend lines that one obtained as the best fitting function.

This fact could be one of the reasons why a considerable variation of the COD fractions, especially regarding S_S was determined when the same wastewater was investigated twice. However, due to the construction of the respirometer it might also be possible that unnoticeable air bubbles occur temporarily beneath the DO probe which could have affected the measurements. This phenomenon was detected during an operation test with tap water and could not be remedied.

With the aim to reduce the variation of S_S and X_S results when the same wastewater was investigated twice, a statistical analysis was carried out on the OUR measurements. Outlier detection according to the Nalimov test was conducted to identify not reliable measurements (Lozán and Kausch (1998)).

The Nalimov test is applicable to normal distributed functions but the acquired OUR data show an exponential evolution in time, when the substrate was added until the endogenous respiration was reached. Hence, the OUR data distribution was analysed again as it was done to determine the slowly and readily biodegradable substrate (see Figure 14). Thereby, the exponential part of the OUR data was transformed logarithmically to obtain a linear function. Then the slope of this function was determined and eliminated to eventually obtain a normal distributed data set where the measurements can be compared to each other. Certainly, the logarithm was not applied to the linear part of the OUR measurements; only the slope had to be eliminated.

According to equation (7) γ^* is calculated for the outlier suspected measurement x^* and is being compared to a corresponding table value $\gamma(95)$, $\gamma(99)$ and $\gamma(99.9)$ (Kaiser and Gottschalk (1972)) dependent on the degrees of freedom respectively the number of measurements. Is γ^* equal or greater than γ(95) (see table in Appendix B) the tested measurement is probably an outlier. For γ* between γ(99) and γ(99.9) this value is significantly an outlier and for γ* greater than γ(99.9) it can be highly significant considered to be an outlier. The standard deviation is located in the denominator of the γ*-equation so that a relatively high standard deviation (as it was determined for the different data sets) has a considerable influence on the γ* calculation. Hence, the 95% confidence (γ(95)) was chosen to be sufficient for the outlier identification.

$$\gamma^* = \frac{\left|x^* - \bar{x}\right|}{s} \cdot \sqrt{\frac{n}{n-1}} \tag{7}$$

Where: x^* .. Measurement
$\bar{x}$ .. Arithmetic mean
s .. Standard deviation
n .. Number of measurements

After the outlier detection, the respective values were replaced by the mean of the immediate neighbourhood which resulted from two measurements before and after the suspected outlier value. This method was applied to keep the main characteristic of the measurement distribution since it was found that a “global” mean would totally smooth the OUR distribution (see Figure 15 and Figure 16). The detection and replacing procedure was conducted so many times until no more outlier was detected.

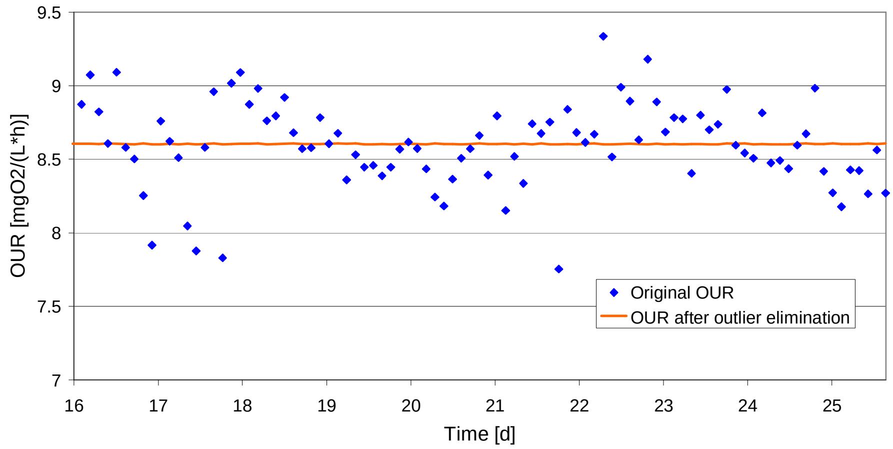


FIGURE 15 EXAMPLE OF THE NALIMOV (OUTLIER) TEST APPLICATION AND REPLACING OF DETECTED VALUES BY USING THE MEAN OF THE WHOLE ENDOGENOUS RESPIRATION OUR DATA (WASTEWATER FROM 15.30 P.M., SECOND RESPIROMETRIC MEASUREMENT)

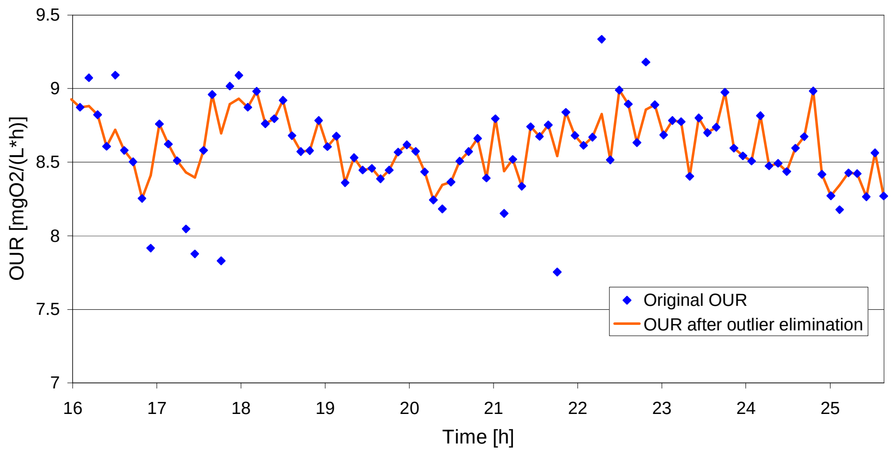


FIGURE 16 EXAMPLE OF THE NALIMOV (OUTLIER) TEST APPLICATION AND REPLACING OF DETECTED VALUES BY USING A NEIGHBOURHOOD MEAN RESULTED FROM TWO MEASUREMENTS BEFORE AND AFTER THE OUTLIER SUSPECTED VALUE (WASTEWATER FROM 15.30 P.M., SECOND RESPIROMETRIC MEASUREMENT)

As the OUR data were cleaned the slowly and readily biodegradable substrate were determined again and compared to the results that originate from the original OUR data sets. Since the idea to obtain normal distributed measurements from an exponential OUR course can be considered a little extraordinary, it is interesting to verify if the "new" calculated results of the same wastewater show less variation.

In Figure 17 it can be recognized that in contrast to the expected outcome, the standard deviation (as an indication for the variation between two respirometric tests analysing the same wastewater) of slowly biodegradable substrate is approximately 2.3 times higher when the outliers are eliminated. But regarding the readily biodegradable substrate it can be observed that by cleaning the OUR data a 70% minimisation of the standard deviation can be achieved. Furthermore, a significantly lower S_S concentration (S_S original: 61 mgCOD/L → S_S cleaned up: 36 mgCOD/L) was calculated by using the data without outliers. This highlights the previous prediction concerning the sensitivity of the results of S_S calculation, especially due to altering the slope arising from the degradation of slowly biodegradable substrate (see Figure 14).

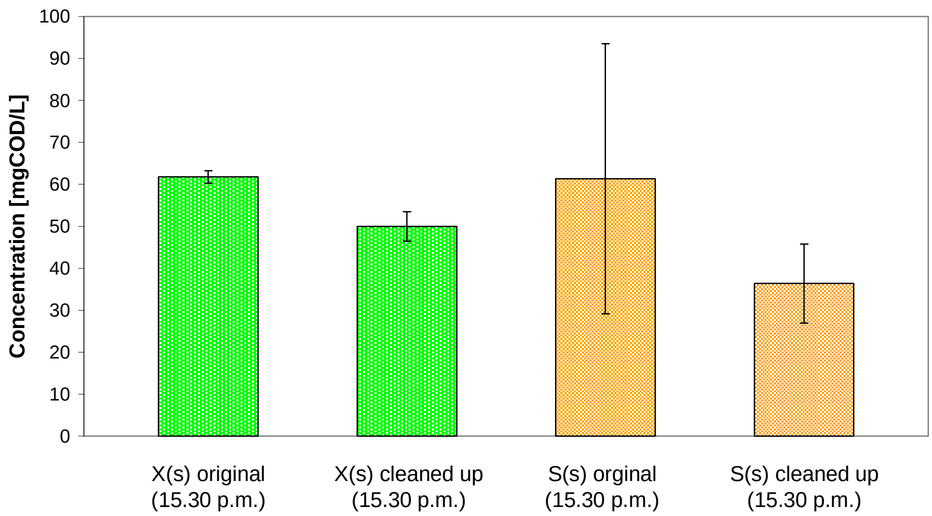


FIGURE 17 COMPARISON OF THE BIODEGRADABLE SUBSTRATE FRACTIONS (AND THEIR STANDARD DEVIATION) RESULTING FROM ORIGINAL OUR DATA (WASTEWATER 15.30 P.M.) AND OUR DATA WHERE OUTLIERS WERE ELIMINATED.

Similarly, results were also examined for the two other (7.30 a.m. and 12.00 p.m.) wastewaters. The example from 15.30 p.m. was chosen to provide a good illustration for the whole problem. Concluding it can be said that an analysis and interpretation of OUR data by using outlier tests was renounced since an overall minimisation of the standard deviation could not be proven. Even if a lower variation between the two results from the same wastewater was calculated (as was the case for S_S in Figure 17) this statistical comparison of only two measurement tests is very risky. This means that no improvement due to outlier tests can be observed with reliable statistical certainty. Thus, it was not possible to explain the range of the substrate measurements to the true value. However, it is highly certain that the OUR measurements are results of real processes and as long as reliable statistic analysis can't be applied, all measurements and therewith the characteristic of the data set have to be maintained.

To verify the determined results for plausibility and to obtain a more detailed characteristic of the different wastewaters a COD balance according to the ASM (Activated Sludge Model) No. 2 (Gujer et al. (1995)) was developed. The total oxygen demand which can be easily and relatively quickly analysed is a sum of biodegradable and non-biodegradable organic matter (See equation (8)). Thus, next to S_S (which is a sum of S_A and S_F) and X_S there is also active heterotrophic biomass X_H that is considered as a further part of the total COD. Furthermore, X_I (inert particulate organic material) together with S_I (inert soluble organic matter) are non-biodegradable fractions of the total COD (Henze et al. (2000)). Due to the limited data availability at the Saint Nicolas treatment plant all the mentioned COD parts had to be estimated. That is, they had to be calculated or reasonably assumed by using literature values as explained in the following.

$$COD_{Tot} = S_A + S_F + S_I + X_I + X_S + X_H \qquad \text{(Henze et al. (2000))} \qquad (8)$$

Where:
- S_A .. Volatile acids (acetate)
- S_F .. Readily (fermentable) biodegradable substrate
- S_I .. Inert, (soluble), non-biodegradable organics
- X_I .. Inert, (particulate), non-biodegradable organics
- X_S .. Slowly biodegradable substrate
- X_H .. Heterotrophic biomass

The inert soluble organic matter S_I could be obtained by taking the difference between the analysed soluble COD and the readily biodegradable substrate S_S since S_I was the only soluble part of the total COD that was not examined. Indeed, the analysed soluble COD may also include some coagulates that can pass filter paper with a pore size smaller than 1.5 µm. These coagulates actually belong to the particular COD fraction but this discrepancy is estimated to be negligible in the context of the simplified COD balancing.

As the COD balance shows, only heterotrophic biomass is taken into account because autotrophic biomass as well as bacteria responsible for phosphorus removal only constitute a negligible part of the total COD. Henze et al. (2000) gives a typical range for X_H which is between five and ten percent of the COD for typical municipal wastewater. Regarding the relatively small catchment and therewith the quite short distance that has to be passed by the wastewater a high biomass development in the sewer system can be excluded. In this case, a significant variation during the course of a day can also be excluded. Hence, the heterotrophic biomass is estimated to be ten percent of the total COD for each wastewater sample.

Finally, the remaining unknown COD fraction X_I can be obtained by the difference between the total COD and the Sum of all other COD parts.

TABLE 11 COD BALANCE OF THE INVESTIGATED WASTEWATERS BY USING SPECTROPHOTOMETRIC AND RESPIROMETRIC ANALYSIS EXCEPT FOR X_H AND X_I WHICH WERE OBTAINED THROUGH LITERATURE RECOMMENDED COD RATIOS (TOT COD, SOL COD X_S, S_S, X_H, S_I, X_I [mgCOD/L])

Time	X_S	X_S/Tot COD	S_S	S_S/Tot COD	X_H	X_H/Tot COD	S_I	S_I/Tot COD	X_I	X_I/Tot COD	Tot COD	Sol COD
7.45 a.m.	49	0.26	29	0.16	19	0.10	31	0.17	58	0.31	185	60
12.00 p.m.	90	0.29	63	0.20	31	0.10	58	0.19	70	0.22	312	121
15.30 p.m.	62	0.26	61	0.26	23	0.10	35	0.15	53	0.23	234	96
Reference ratio*		0.30 - 60		0.12 - 0.30		0.05 - 0.15		0.05 - 0.10		0.10 - 0.15		

* Reference range for typical municipal wastewater (primary effluent) according to Henze et al. (2000)

In Table 11 the different COD balances of the analysed wastewater samples are shown.

In general, it has to be pointed out that the non-biodegradable organic material (S_I and X_I) is determined to be much higher than the reference range Henze et al. (2000) recommend. This is due to two factors: First, the slowly biodegradable substrate exhibits a low concentration for all wastewaters as mentioned before. Second, the heterotrophic biomass is assumed to be a constant fraction of the COD. Hence, the non-biodegradable COD fractions present inevitably a large part of total organic material. The highest fraction of X_I is reached in the morning and shows a relatively constant COD ratio with respect to samples taken at noon and in the afternoon. The inert soluble organic material appears to be relatively constant throughout all wastewater investigations.

4.5 Conclusion

The investigated wastewater samples taken at the Saint Nicolas WWTP were analysed twice respirometrically and several times spectrophotometrically (which was less extensive) not only to obtain the biodegradable organic concentration but also to get an idea about the general wastewater composition. It was found that the results regarding slowly and readily biodegradable material of the same wastewater investigated under equal laboratory conditions shows a very significant variability. This was especially true for the S_S COD fraction. Due to snowmelt the investigated wastewater showed a considerable dilution for both the organic material as well as the ammonia concentration, compared to the results of the respirometer pre-tests. Because of this, the graphical determination of the X_S and S_S fractions turned out to be relatively complex due to the lower Oxygen Uptake Rates.

Moreover, considerable measurement noise on the respirometric data was identified. It can be explained by the setup of the respirometer where small air bubbles can occur beneath the DO sensor. With the aim to reduce the noise and, therewith, the variation of the respirometric measurements from the same wastewater an outlier test was carried out to identify and replace suspected values. However, a general reduction of the measurement noise could not be observed by eye and also not be proven statistically. Thus, it was decided to analyse and interpret the obtained OUR data without outlier elimination and to accept the investigated variations.

As a result of the laboratory investigations it can be concluded that a significant fluctuation (maximal 10%) of the readily biodegradable substrate fraction of the total COD was observed. In Figure 18 an increase of the S_S fraction (from 16% S_S in the morning until 26% of readily biodegradable substrate in the afternoon) can be noticed. In contrast, the variation of the slowly biodegradable substrate fraction from 26 % in the morning and afternoon and 29 % at noon is negligible.

Moreover, the spectrophotometric measurements confirm the observed fluctuation in biodegradable substrate since the soluble part of the total COD shows a similar course of COD ratio for the three wastewater samples. The variation does not seem to be as significant as the S_S / COD ratio. One explanation could be that some coagulates may have passed the filter paper that was used to analyse the soluble COD fraction. Nevertheless, the similar trend supports the plausibility of the respirometric results.

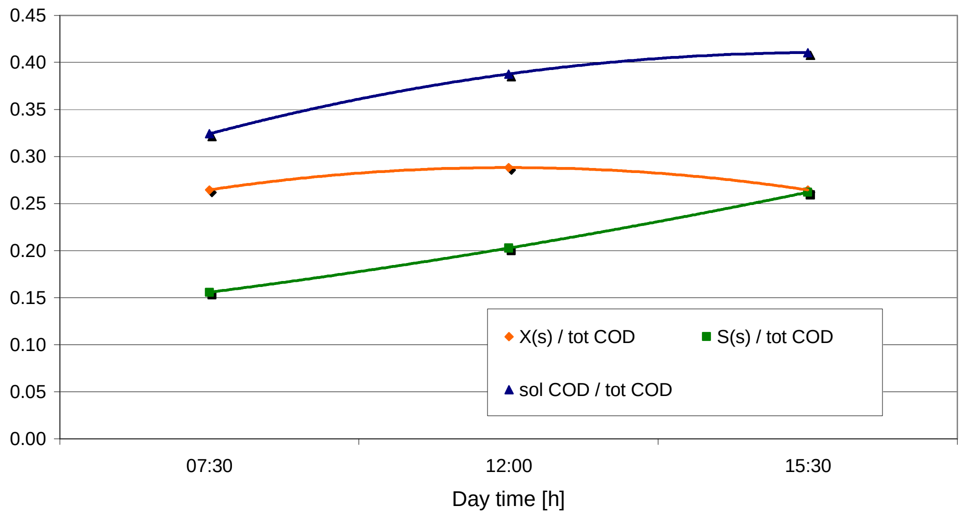


FIGURE 18 READILY AND SLOWLY BIODEGRADABLE SUBSTRATE FLUCTUATION IN RATIO TO THE TOTAL COD

In conclusion, a fluctuation of the COD fraction in the form of readily biodegradable substrate during dry weather conditions (taking into account some influences due to snowmelt) was respirometrically confirmed. A general effect on the treatment performance of a WWTP based on the presented results can be expected since changes of this COD fraction would mainly have an influence on the denitrification and biological phosphorus removal at the treatment plant.

The relatively small size of the catchment of the Saint Nicolas WWTP and the resulting short flow times of the sewage may limit the development of a considerable hydrolysis process so that only the dischargers of the sewer system could influence the X_S / total COD ratio during dry weather flow.

Ahnert et al. (2005) describe that besides construction and size of sewer systems also temperature is one of the most important influencing factors on biological process and explicate that a considerable difference of wastewater quality (taken into account COD, TKN and TP) at the WWTP inlet during winter and summer was observed. For example, the nitrate concentration was determined to be 80% lower at 17 degrees than at 8 degrees as investigations at the Dresden-Kaditz WWTP (Germany) under different temperatures revealed. The discovered highly significant reduction of organic matter, nitrogen, and phosphate during summer was explained with increasing in-sewer degradation of biodegradable matter. Consequently, the outside temperature which lay between -5 and 6 degrees during the sampling at the Saint Nicolas WWTP has to be considered since this fact would result in low biological processes in the sewer system.

5 INVESTIGATION OF THE BLEESBRUCK SEWER MODEL WITH RESPECT TO BIODEGRADABLE SUBSTRATE PREDICTION

5.1 INTRODUCTION

This thesis is mainly based on the dissertation of Solvi (2007) where an integrated model (consisting of sewer, wastewater treatment plant and river) was developed and implemented in the software WEST® and further calibrated by using data from different measurement campaigns and data directly available from the WWTP. The assumptions made in chapter 4 that the biodegradable substrate fractions can fluctuate in ratio to the total COD (due to hydrolysis) during the course of a day will be implemented in the Bleesbruck sewer model. Furthermore, impacts on the effluent quality will be investigated by analysing different simulations carried out with WEST®. First, the WEST® software, the WWTP and the sewer model characteristics and dynamics will be introduced to provide a good basis for the interpretation of the subsequent simulation results.

5.2 MODELLING AND SIMULATION TOOL WEST®

Common state of the art modelling and simulation packages to replicate wastewater treatment can be classified in three different types. First, general simulators like Matlab/Simulink or ACSL can be used. Second, closed simulation software like Biowin and STOAT are specially designed to model wastewater treatment plants where only predetermined models can be used. And finally, open source software like Simba, GPS-X and WEST® are also dedicated to WWTP simulations but here the user is able to implement his own mathematical model in case the models already available in the software do not conform the individual needs (Meirlaen (2002)).

The software package WEST® was developed by the Department of Applied Mathematics, Biometrics and process Control (Biomath) based at Ghent University, Belgium and Hemmis N.V. (Kortrijk, Belgium; an environmental IT development company). Today WEST® has been taken over by the newly formed Belgian company MOSTforWATER (Kortrijk, Belgium), an alliance between Hemmis, Ghent University, VINNOF, Baekeland Fund II and key employees.
The software is basically applied for evaluation of design options, training process optimisation, development of control strategies, evaluation of operation strategies, operator support and real-time control. WEST® presents a user friendly software platform and cannot only be applied to WWTP

modelling but also to simulation of any other process and unit (sewer system, river) of an urban wastewater system that can be described as a structured collection of Differential and Algebraic Equations (DAEs). Moreover, it is also possible to simulate the whole urban wastewater system in an integrated view.

Basically, three main environments can be found in WEST®.

In the Model Editor environment the user is allowed to modify and to add models which are described in the dedicated Model Specific Language (MSL-USER),a high level object-oriented language specifically developed to incorporate models. The model base which has a hierarchical structure is made for maximal reuse. All reusable knowledge (e.g. mass balance and default parameters) are globally defined and can be reused by a professional user to build a new model.

All biological processes (based on Activated Sludge Models) their components, stoichiometry and kinetics are presented in the Pertersen Matrix Editor where the user can easily create and integrate the Petersen Matrix into the modelbase.

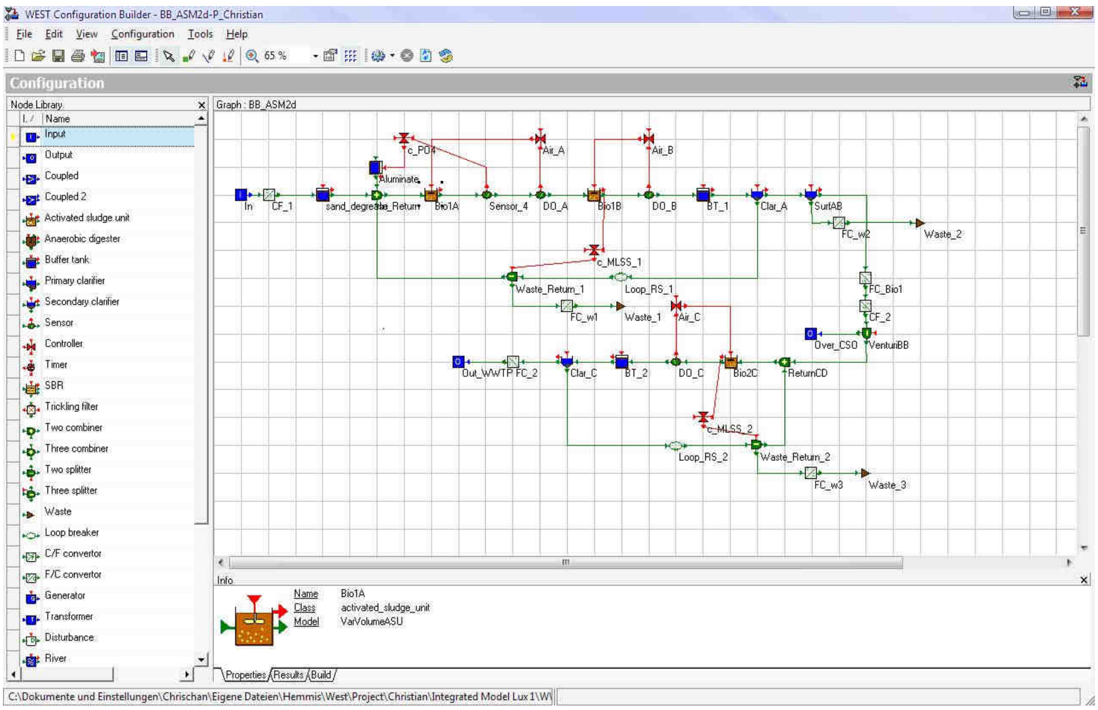


FIGURE 19 ILLUSTRATION OF THE CONFIGURATION ENVIRONMENT OF WEST®

In the Configuration Environment the user constructs the desired system (for example a WWTP) by choosing different subcomponents (like clarifier, activated sludge tank etc.) by drag and drop from the left hand side of the interface (see Figure 19). For each unit the user can select among a series of models that are coded inside the model base. WEST® includes the most recent mathematical models, for example processes in activated sludge tanks (ASM1, ASM2, ASM2d, ASM3, ASM3 Bio-P) as well as models for trickling filters, sand filters, settling tanks and anaerobic digestion reactors (ADM 1). After the system has been built the model will be converted into MSL-USER that is automatically added to the model base for further use in new model configurations. Next, the MSL-file is parsed by WEST® into low-level (C^{++}) code that is compiled to a WEST® model library file that can be loaded into the Experimentation Environment.

The Experimentation Environment enables the user to run simulations ("virtual" experiments) where parameter values can be manually changed. Also, automatic parameter estimation, scenario analysis and sensitivity analysis can be performed with the model. Moreover, the experimentation environment offers graphical analysis at different points of the system configuration (see Figure 20) as well as generating external file output where simulation results can be further investigated with spreadsheet programs like Microsoft Excel (See Figure 20).

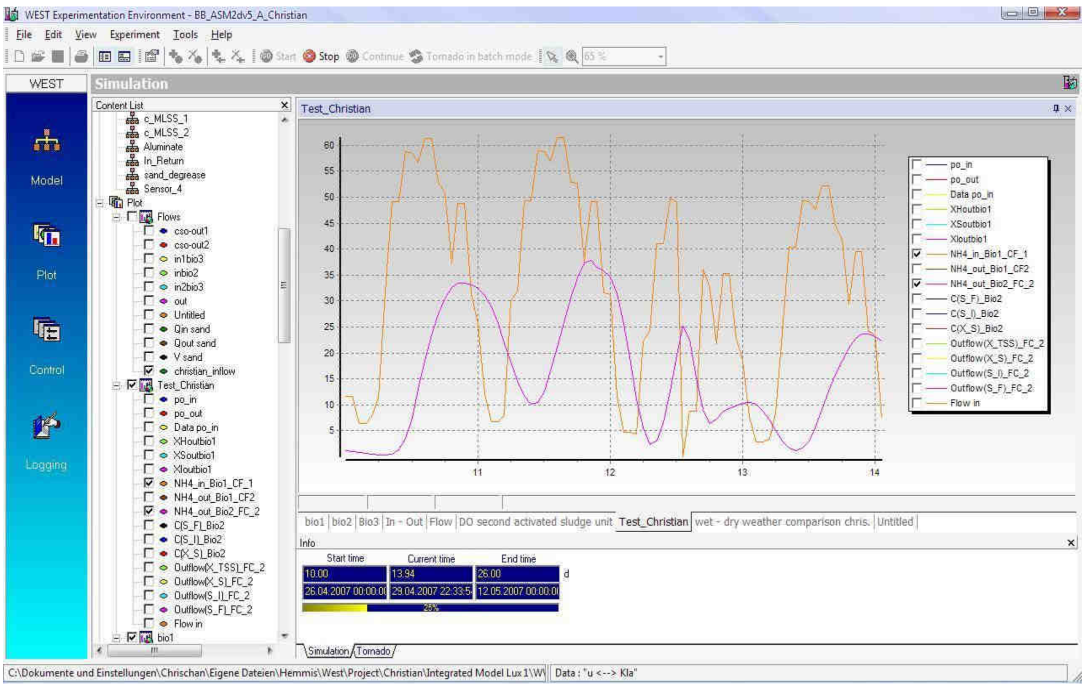


FIGURE 20 ILLUSTRATION OF THE EXPERIMENTATION ENVIRONMENT OF WEST®

5.3 The Bleesbruck WWTP model construction

In the following the sewer and WWTP model which was implemented within the work of Solvi (2007) will be presented. The model implementation and calibration was part of the Bleesbruck case study that was already described in chapter 3.4. The Bleesbruck catchment comprises around 24 000 habitants while wastewater of 10 000 PE (population equivalents) is discharged by industry.

First, a brief overview will be given of the main plant characteristics to give the reader a better understanding of the overall subject matter. Afterwards, the model describing the biological processes will be analysed more closely and the different considerations of pollutants in the sewer model in relation to the WWTP model will be pointed out by describing the interface between both systems.

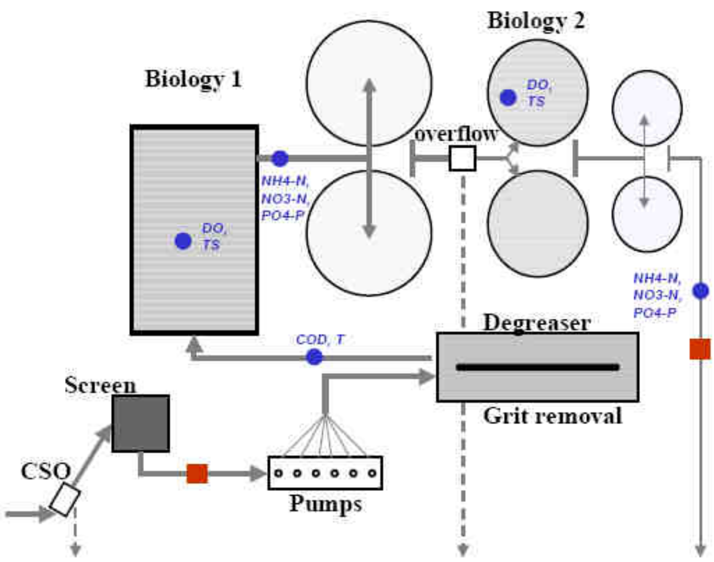


FIGURE 21 WASTEWATER FLOW SCHEME OF THE BLEESBRUCK WWTP. MARKED IN BLUE CIRCLES ARE THE ONLINE MEASUREMENT LOCATIONS AND MARKED IN RED SQUARES ARE THE CD4WC[2] MEASUREMENT POINTS FOR TSS, COD, NH_4^+, NO_3^-, AND PO_4^{3-}. THESE DATA WHERE USED TO CALIBRATE THE WWTP MODEL (SOLVI, 2007)

[2] The Cost-effective development of urban wastewater systems for Water Framework Directive compliance is a research project supported by the European Commission.

The Bleesbruck wastewater treatment plant consists of several components (see Figure 21) like screen, grit removal/degreaser unit, a first stage activated sludge (AS) unit where the organic material is removed, two clarifiers that are modelled as point settler and a second stage with two AS tanks for nitrification and two other clarifiers. Phosphorus is eliminated by chemical precipitation. Two centrifuges provide for sludge dewatering. Furthermore, two digesters are also installed on-site and provided with sludge from neighbouring WWTPs to make the overall operation efficient. The WWTP has been modernised with tools for real-time monitoring and control of the treatment processes (Schosseler et al. (2000) and Schosseler et al. (2003)) to achieve a higher treatment efficiency. A model that is currently used to regulate the second biology was implemented into the SIMBA software (ifak System GmbH; Magdeburg, Germany).

After heavy material and solids are removed in the mechanical treatment stage, the raw water reaches the activated sludge unit where organic carbon and ammonia nitrogen is removed through bio-chemical processes.

As it was already pointed out in chapter 5.2 the software WEST® provides a large number of models that can be used to represent the dynamic processes of the treatment plant. For the integrated study of Solvi (2007) the ASM (Activated Sludge Model) No.2d (Henze et al. (1999)) has been chosen to simulate the biological processes and the presence of phosphorus removal by chemical precipitation. Indeed, the ASM No.2d includes variables and processes of biological and chemical phosphorus removal (Solvi (2007)). The activated sludge approach of Henze et al. (1999) is the most common model to predict biological treatment processes and offers a good compromise between simplicity and complexity.

The ASM No.2d is an upgraded version of the activated sludge model number two that has been developed since one found out that polyphosphate accumulating organisms (PAO) grow also under anoxic conditions and contribute to denitrification and phosphorus removal in the anoxic tank (Henze et al. (1999)). The ASM No.2d contains 19 state variables which represent concentrations for: heterotrophic and autotrophic biomass, different kinds of organic matter as well as inorganic pollutants. 21 biological processes are considered within the model that range from aerobic and anoxic growth of microorganisms, over anoxic and aerobic storage of polyphosphate, lysis, fermentation, precipitation to redissolution. A matrix format (developed by Petersen (1965)) is used to summarize the multitude of interactions between wastewater components and the processes in the activated sludge tank in an understandable way (Henze et al. (1999)).

Figure 22 shows a very simplified view on the Petersen matrix where wastewater compounds are confronted with different processes and their kinetics. The depicted compounds and processes are only an extract of the whole approach and should only illustrate the relations between the compounds. The stoichiometric factors that are normally used (in Figure 22 simplified with plus → increase and minus → decrease of concentration) specify how much of each wastewater component is used for the respective process whereas the kinetic equation accounts for the rate of each process. Thus, as an example for aerobic growth of heterotrophs (depicted as BM1) involves the consumption of organic substrate and reduction of oxygen. Concluding, the reaction rate $r_{i,j}$ results from equation (9).

$$r_{i,j} = \nu_{i,j} \cdot \rho_j \qquad (9)$$

Where: $\nu_{i,j}$ ..stoichiometric coefficient [e.g. $gCOD_{SS}\ gCOD_{XBM}^{-1}$]
ρ_j ..process rate [e.g. $gCOD_{BM}\ m^{-3}d^{-1}$]
$r_{i,j}$ ..reaction rate [e.g. $gCOD_{SS}m^{-3}d^{-1}$]

compounds i / **processes j**	O_2	org.C	N_2	NO_3	NH_4	BM1	BM2	**process rate ρ_j**
aerobic growth	–	–				+		$\rho_{aer\ growth}$
decay	–					–		ρ_{decay}
denitrification		–	+	–		+		$\rho_{denitrification}$
nitrification	–			+	–		+	$\rho_{nitrification}$
	stoichiometry (how much)							kinetics (how fast)

FIGURE 22 PRINCIPLE OF THE PETERSEN MATRIX THAT IS USED TO REPLICATE THE ACTIVATED SLUDGE PROCESSES

According to this principle (Figure 22) the biological processes of the Bleesbruck wastewater treatment plant were implemented in WEST®.

In Figure 23 it can be seen that the first biology is modelled as two bio-units. It had been found out that the use of two activated sludge models to replicate one biological tank is an appropriate way to model the hydraulics of the first AS stage (Schosseler et al. (2003)). The oxygen concentration respectively the aeration within the activated sludge tanks is controlled by PI (proportional-integral) controllers. There, the user enters a suitable setpoint for the oxygen concentration that is to be provided by the aeration device.

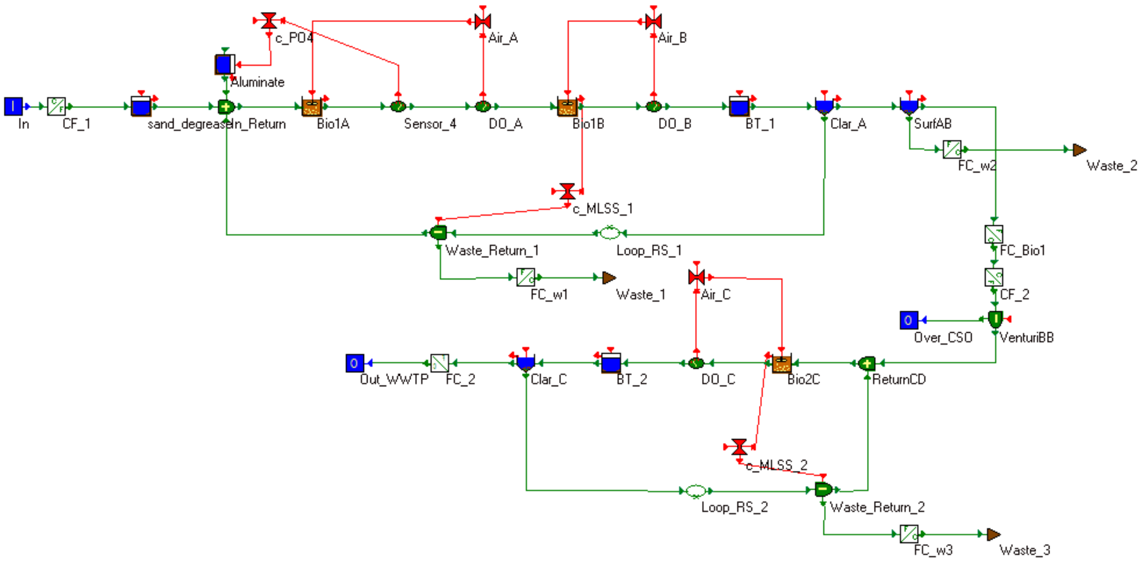


FIGURE 23 BLEESBRUCK WWTP MODEL ILLUSTRATED IN THE EXPERIMENTATION ENVIRONMENT OF WEST® (SOLVI, 2007)

The WWTP model was calibrated by setting aeration control parameters so that the dissolved oxygen concentration behaviour could be reproduced. Afterwards, the solids balance was checked and pollutants concentrations were tried to be adjusted to reflect the online measurements and collected data from measurement campaigns (Solvi (2007)). The best fitting behaviour of the biological processes could be found through adjustments of some kinetic parameters in the Petersen matrix (Henze et al. (1999): growth rate of heterotrophs, rate for lysis and decay of heterotrophs, growth rate of autotrophs and reduction factor for denitrification.

5.4 The Interface between The Bleesbruck sewer system and the WWTP

As pointed out in chapter 4.4.4.2 the COD balance of the ASM No.2 respectively 2d model considers soluble and particulate organic matters. Both COD fractions are subdivided into biodegradable and non-biodegradable organic material. The non-biodegradable part of particulate organic matter is furthermore split into biomass and inert organic matter. In the sewer system only soluble and particulate organic material is taken into account. Hence, an interface between both systems has to translate the soluble and particulate organic material coming from the sewer into the ASM No.2d components. Table 12 shows how transfer factors convert the measured COD parameters that originate from the outflow of the sewer model to ASM No.2d based COD fractions (Solvi (2007)). The conversion of organic material by the sewer-WWTP interface is an appropriate way to construct integrated models especially in view of the computing time and the model complexity that would be much higher when ASM components would also be considered in the sewer system.

TABLE 12 INTERFACE BETWEEN THE BLEESBRUCK SEWER AND WWTP MODEL CONSIDERING THE ORGANIC MATERIAL

Measured variable	ASM2d variable	f_{trans}	unit
COD soluble	S_F	0.48	$gCOD/m^3$
	S_A	0.32	$gCOD/m^3$
	S_I	0.2	$gCOD/m^3$
COD part. =COD total - COD soluble	X_S	0.8	$gCOD/m^3$
	X_I	≈ 0.04	$gCOD/m^3$
	X_H	0.16	$gCOD/m^3$
	X_{TSS}	0.75	$gTSS/m^3$ $gCOD/m^3$

Regarding ammonia as well as ortho-phosphate coming from the sewer system it is to be mentioned that here no transformation functions are necessary so that the sewer outflow equals the WWTP inlet.

5.5 THE BLEESBRUCK SEWER MODEL CONSTRUCTION AND CONFIGURATION

The drained catchment area investigated by Solvi (2007) measures approx. 900 ha of which ca. 20 % can be considered as impervious. The combined sewer network has a length of 60 km.

Within the work of Solvi (2007) the KOSIM modelling tool (designed for long-term simulations of dry weather generation, surface runoff and transport in the sewer system; ITWH (2000)) was considered to be an appropriate tool that could be transformed into a lot of ordinary differential equations (ODEs) and implemented into WEST® to represent the Bleesbruck catchment. Due to the limited computing time which was essential for the long-time simulation that were conducted, this conceptual model appeared appropriate. KOSIM Software can be applied to evaluate combined sewer systems and determine pollutant loads from the sewer. KOSIM allows taking into account up to 50 combined sewer overflows. Not all characteristics of the KOSIM model have been translated into WEST® but only those that were deemed to be necessary to simulate the catchment in an integrated view. The KOSIM model is able to give up to five pollutant loads in response to individual rain events and consists of different submodels which are hydrological nature.

The sewer model considers three different kinds of pollutants which are chemical oxygen demand (COD), nitrogen and phosphorus. These components are further subdivided into soluble and particulate components. According to Figure 24 the sewer system can be divided into the catchment model and the sewer transport model which are linked together. With respect to the catchment model all sewage producing units (rain, households and industry) and their properties like evaporation, pervious/impervious area, day time etc. are taken into account to generate the wastewater flow that enters the sewer system. The sewer transport model is a hydrological model where the wastewater flow is described by assuming the dynamic flow process as a linear tank cascade. Here, the properties of the wastewater flow are mostly influenced by the sewer construction effecting, for example, retention, back water effects and CSO (combined sewer overflow) events. Regarding the wastewater quality only sedimentation and resuspension of particulate compounds are considered.

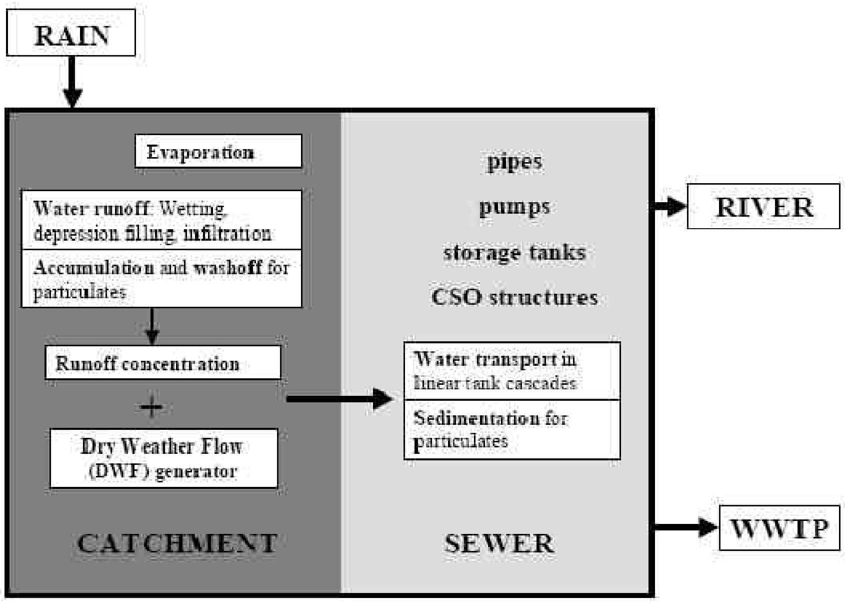


FIGURE 24 ELEMENTS AND PROCESSES WITHIN THE KOSIM – WESTSEWER MODEL (SOLVI (2007))

5.5.1 THE BLEESBRUCK URBAN CATCHMENT MODEL

The urban catchment comprises all the wastewater producing units like households and industry with their local sewer system including the area surrounding these units. The municipal wastewater enters, together with rainwater, into the main collector of the sewer system. Thus, all the processes beginning with rain over run-off, water losses, accumulation and wash-off effects as well as municipal wastewater production have to be modelled for the urban catchment unit (see Figure 24).

Rain input data can be implemented for each subcatchment in WEST® with a simple time-rain vector format. The rain is subject to evaporation that is dependent on the season of the year and the annual mean evaporation specific to that region. The potential evaporation for a specific time of the day is taken into account by hourly factors in the catchment model. The effective rain entering the main collector system is determined by a model transforming rain into run-off. The area connected to the sewer system as well as the portion of pervious and impervious area play a significant role regarding the losses due to wetting, depression and infiltration. A runoff concentration model accounts for the time of travel and retention of peaks by assuming the catchment to be a linear reservoir. Finally, the runoff is added to the dry weather flow which is generated by a further submodel (see Figure 25).

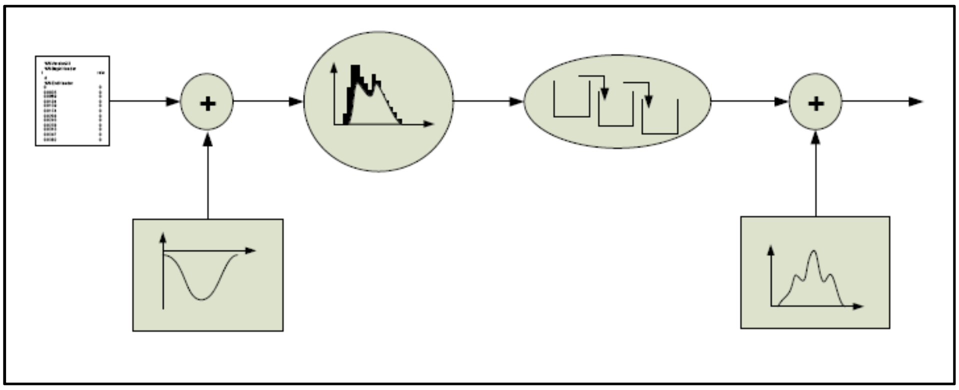


FIGURE 25 PRINCIPLE OF THE FLOW (DWF) GENERATOR CONTAINING DIFFERENT SUBMODELS. FIRST, THE INCOMING RAIN DATA IS TRANSFORMED INTO EFFECTIVE RUNOFF (I.E. SUBTRACTION OF ANY LOSSES ON SURFACE). AFTERWARDS THE RUN-OFF UNDERGOES TIME TRANSLATION AND RETENTION AND IS THEN ADDED TO THE DWF. (SOLVI (2007))

Due to the aspired simplicity of the integrated model and the needed minimisation of the number of parameters, non-water components are supposed to run off from impervious surfaces only. The diurnal pollutant load that is originating domestically is dependent on consumer habits e.g. the shower that is often taken in the morning and night. Furthermore, it can be generalized that most of the household activities (for example doing laundry) is performed off the job. Thus, a different water use during a weekday compared to Saturday and Sunday has to be taken into account by hourly factors as well as coefficients which account for the weekend effect. Moreover, a seasonal effect due to additional tourism pollution can be observed and has to be considered too.

The simplest modelling approach to calculate the flux $F_n(t)$ of pollutants n at time t is determined below (equation (10)).

$$F_n(t) = \overline{C_n} \cdot Q(t) \tag{10}$$

Where $\overline{C_n}$ represents the mean pollutant concentration which is defined by the user, and flow $Q(t)$ which is generated from the model by using rain data. The WEST® sewer-generator model, a submodel of the Bleesbruck catchment model, takes into account all the above mentioned influences on domestic waste production plus the impact through industrial wastewater by the use of hourly, daily and monthly patterns.

```
CLASS SewGenerator
(* class = ""; category = "" *)
"A generator model"

OBJ Outflow (* terminal = "out_1" *) "Outflow" :
OutSewTerminal := {: causality <- "COUT" :};

OBJ Mean_S "Vector containing the mean concentration" :
SewConcentrationVector := [{:value <-1000000 :},
{:value <- 550 :}, {:value <- 70 :}, {:value <- 12 :},
{:value <- 70 :}, {:value <- 65 :}, {:value <-10 :};];
OBJ SewSpecificVolume (* hidden = "1" *) : SewSpecificVolumeVector;};

interface.Outflow[H2O_sew] = - parameters.Mean_S[H2O_sew] * state.FlowHelp;

{FOREACH s_comp_index IN {2 .. NrOfSewComponents}:
interface.Outflow[s_comp_index] = - parameters.Mean_S[s_comp_index] *
state.PollutionHelp;}
```

FIGURE 26 EXTRACT OF THE BLEESBRUCK SEWER-GENERATOR SUBMODEL IMPLEMENTED IN WEST®

As shown in the WEST® MSL model code of Figure 26 the considered sewer components (highlighted in different colours) water (cyan), particulate COD (yellow), particulate nitrogen (red), particulate phosphorus (magenta), soluble COD (green), ammonia nitrogen (grey) and ortho-phosphates (brown) are defined by the mean concentration in the sewer-concentration-vector where a unit of mg/L is used. Regarding the water component one rather speaks about density which can be simply assumed to be 1,000,000 mg/L. After this declaration, the water flow is related to the hourly flow pattern ("state.FlowHelp" according to Figure 26) that represents the dry weather flow fluctuation during the day. The non-water components are multiplied by an hourly pollution pattern ("state.PollutionHelp"). All patterns that include monthly, daily and hourly dependencies regarding the wastewater composition and flow rate are declared in a further submodel, called DWF-generator.

In the DWF generator (Figure 27) different flow and pollution patterns are implemented. The intensity of the DWF fluctuation is dependent on the population density as well as on potential industrial influences. To represent the individual dry weather flow of a sewer system the user can choose the best fitting default pattern in the WEST® Experimentation Environment or he can change the pattern directly in the MSL-file. This dual approach reveals a very useful option of WEST® where the user is able to create an individual model in the programming code through the "open source" property. Default values are taken from appropriate literature e.g. DWA[3] gives reasonable DWF fluctuations during the day depending on the catchment size and population.

```
CLASS DryWeatherFlow
(* class = "generator"; category = "" *)
"A dry weather flow generator for subcatchments in Kosim"

OBJ SIDENPollution (* hidden = "1" *) "Pollution pattern defined by user" :
HourlyDailyPattern
:= [{: value <- 0.45:},{: value <- 0.5 :},{: value <- 0.45 :},
{: value <- 0.6 :},{: value <- 0.8 :},{: value <- 1.1 :},
{: value <- 1.2 :},{: value <- 1.2 :},{: value <- 1.1 :},
{: value <- 1.05 :},{: value <- 1.1 :},{: value <- 1.3 :},
{: value <- 1.5 :},{: value <- 1.6 :},{: value <- 1.5 :},
{: value <- 1.4 :},{: value <- 1.3 :},{: value <- 1.3 :},
{: value <- 1.3 :},{: value <- 1.1 :},{: value <- 0.7 :},
{: value <- 0.5 :},{: value <- 0.45 :},{: value <- 0.4:};];
```

FIGURE 27 EXTRACT OF THE BLEESBRUCK DWF – GENERATOR SUBMODEL. THE HOURLY POLLUTION PATTERN THAT CAUSES THE DIURNAL FLUCTUATION OF POLLUTANT CONCENTRATION AT THE CATCHMENT OUTLET IS SHOWN.

[3] Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall; German association for water management, wastewater and waste

```
OBJ SIDEN (* hidden = "1" *) "Typical for Bleesbruck WWTP" :
HourlyDailyPattern
:= [{: value <- 0.3:},{: value <- 0.3:},{: value <- 0.3 :},
{: value <- 0.35 :},{: value <- 0.4 :},{: value <- 0.7 :},
{: value <- 1.5 :},{: value <- 1.7 :},{: value <- 1.3 :},
{: value <- 1.1 :},{: value <- 0.8 :},{: value <- 1 :},
{: value <- 1.5 :},{: value <- 1.9 :},{: value <- 1.8 :},
{: value <- 1.5 :},{: value <- 0.8 :},{: value <- 0.7 :},
{: value <- 1.5 :},{: value <- 1.8 :},{: value <- 1.5 :},
{: value <- 0.5 :},{: value <- 0.4 :},{: value <- 0.35 :};];
```

FIGURE 28 EXTRACT OF THE BLEESBRUCK DWF-GENERATOR SUBMODEL. THE HOURLY FLOW PATTERN THAT CAUSES THE DIURNAL FLUCTUATION OF FLOW RATE AT THE CATCHMENT OUTLET IS SHOWN.

To simulate the diurnal wastewater and pollutant fluctuation at the Bleesbruck WWTP inlet an hourly pollution pattern developed by SIDEN[4] was used to relate mean flow and pollutant concentrations to the individual fluctuation characteristics of this specific catchment (as shown in Figure 27 and Figure 28). The hourly pattern is a vector that provides coefficients for each hour of the day to represent variations of the pollution load respectively flow rate. Regarding the Bleesbruck model configuration one assumes that every pollutant follows the same course of the day. The diurnal fluctuation of flow rate follows a separate pattern. The amount of water due to infiltration is modelled as a mean flow per connected area. The intruding water is assumed to be unpolluted and is factorised through a yearly pattern.

5.5.2 The Bleesbruck sewer transport model

The wastewater transport in the sewer system is hydrologically modelled. Different parts of the sewer configuration and their influences on the sewage flow and pollutants concentration are considered. In the following, the sewer transport submodels will be shortly explained with a focus on storage tanks since the fractionation of wastewater pollutants through sedimentation processes in these facilities is of particular interest for this work.

[4] Syndicat Intercommunal de Dépollution des Eaux résiduaires du Nord; local association for wastewater treatment in Northern Luxembourg

The wastewater flow in the pipes is modelled by using individual tank models as linear reservoirs that are placed in series. The user can choose from a series of up to 10 tank models. The Kalinin-Miljukov method is used to divide the real sewer pipes in certain lengths to represent the wastewater flow by the application of the tank cascade model. The pipe diameter, slope and length determine the individual pipe stretches.
The residence time k [d] of such a tank is determined by the equation (11).

$$k = 0.64 \cdot L^{*} \cdot \frac{d^{2}}{Q_{\max}} \tag{11}$$

Where L^* [m] is the corrected specific length of the individual pipe stretches, d [m] presents the diameter of the pipe and Q_{max} [m³/d] is evaluated applying the Colebrooke-White equation (12) (Butler and Davies (2000)).

$$Q_{\max} = a \cdot \left(-2 \cdot \log\left(\frac{2.51 \cdot \upsilon}{d\sqrt{2gds}} + \frac{k_S}{3.71d} \right) \cdot \sqrt{2gds} \right) \tag{12}$$

Where: g .. gravity [m/s²]
v .. kinematic viscosity [m²/s]
k_S .. roughness coefficient
a .. moistened area [m²]

Backwater effects taking place in the sewer system are typically not considered in hydrological models. These effects can especially occur in flat sewer systems with a very rough pipe surface. Also obstructing structures downstream, under dimensioned pipes or accumulation of sewer solids can cause those backflow phenomena. There, the maximum outflow capacity of a pipe is not high enough to conduct all the flow downstream. Thus, wastewater will be stored upstream, i.e. it flows the adverse direction. The backwater effect can induce CSO events that have a significant impact on the river water quality. Hence, it was very important to consider these processes with respect to the integrated view of the Bleesbruck study.
However, a conceptual backflow model with a hydrological approach was developed in WEST® by Solvi (2007). The model is a combiner-splitter combination. Combiner and splitters are submodels without volume. The former adds flows and pollutants coming from different structures. The latter

separates the flow into a set flow (given by the user) and the remaining flow whereas a second splitter type divides the flow into two fractions according to an entered fraction parameter.
The combiner of the backflow model sums the water coming upstream from a tank cascade and the water that is sent back from the downstream backflow model (See Figure 29). The following absolute splitter sends any excess water that exceeds the set point (the maximum flow capacity) of the pipe to the upstream combiner (Solvi (2007)).

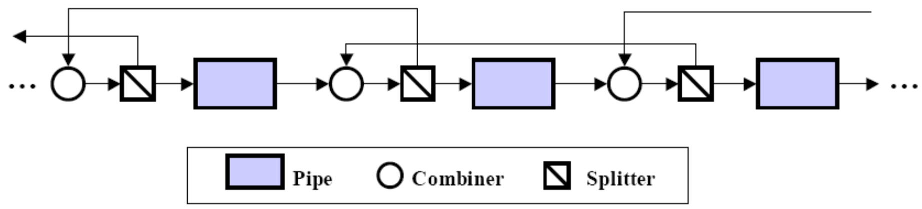


FIGURE 29 PRINCIPLE OF THE BACKFLOW MODEL IMPLEMENTED INTO THE BLEESBRUCK SEWER TRANSPORT MODEL (SOLVI, 2007)

Sedimentation and resuspension behaviour of particulate matter are modelled by considering the approach of Bechmann et al. (1999) which was reformulated in Willems (2004). This grey box model simulates exponential deposition of particulate material. Thereby, the model distinguishes between two different flow conditions (see equations (13) and (14)). A certain dry weather flow $Q_{DWF,max}$ defines the set point and any excess flow causes resuspension of the particulate material that has accumulated before. The more material is available, the more violent the first flush effect will be (Solvi (2007)).

Deposition of particulates during DWF

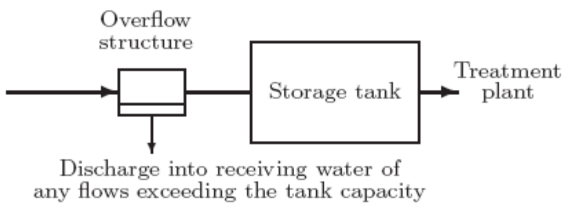
$$\frac{ds}{st} = -\frac{1}{k}(s-\bar{s}) - b_1(Q_{in} - Q_{DWF\max}) \qquad (13)$$

Resuspension of sediment particulates

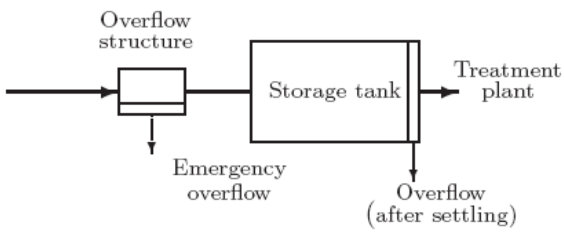
$$\frac{ds}{st} = -\frac{1}{k*}(s-\bar{s}) - b_2(s)(Q_{in} - Q_{DWF\max}) \quad \text{with} \quad b_2(s) = b_{\max}(1-e^{-\frac{s}{k*}}) \qquad (14)$$

Where:	s	.. Deposition / Resuspension matter [g]
	t	.. Time [d]
	k	.. Linear reservoir constant (residence time) [d]
	k^*	.. Deposition rate [1/d]
	$\bar{S}$	.. Mean mass of the considered particulate matter [g]
	b_1	.. Flow influence on sedimentation constant
	b_{max}	.. Flow-mass influence on resuspension constant
	Q_{in}	.. Inflow pipe [m³/d]
	Q_{DWFmax}	.. Maximal dry weather flow [m³/d]

As already mentioned, the river water quality is significantly affected by the amount of wastewater that is discharged directly into the river during heavy rain events. To reduce these volumes different rain retention tanks are installed in the sewer system. For the Bleesbruck sewer transport model two different kinds of storm water tanks are considered (See Figure 30). The tanks can be placed parallel or in series depending on whether the flow is travelling through the tank or only excess water enters the tank during rain events and the tank is emptied when the water level in the pipe allows for it.



(a) By-pass tank: Stormwater tank retaining the first flush of stormwater with immediate overflow when full



(b) Pass-through tank: Stormwater tank with overflow for settled combined sewage

FIGURE 30 TWO KINDS OF STORMWATER TANKS ARE USED IN THE BLEESBRUCK SEWER TRANSPORT MODEL (SCHÜTZE ET AL. (2002)).

According to Figure 30 one also distinguishes between two kinds of construction. The by-pass tank is used to reduce the first flush effect whereas the more contaminated wastewater is retained during rain events to release the WWTP. For this reason the appropriate submodel is named “first flush tank” which is realized in the Bleesbruck sewer transport model. The tank is considered as a fully mixed basin with immediate overflow through an upstream splitter when the maximum retention capacity is reached. The by-pass tank is often used in a smaller catchment where flow times can be determined below 15 – 20 minutes.

In contrast, pass-through tanks (also called stormwater tanks) are generally built in larger catchments where first flush effects are more important than the concentration distribution of pollutants over time. These retention tanks treat the wastewater mechanically by sedimentation before excess water is discharged into the river. The stormwater tank is of special interest within this work since the sedimentation factor is situated between 0 and 1 which gives the user a relatively easy access to fit the particulate compounds to the measured data.

```
CLASS StormwaterTank
(* class = "retention_basin"; category = "" *)
"A tank with lateral weir and overflow when full"

OBJ sedimentation "Sedimentation fraction" :
Fraction:= {: value <- 0.25:};

{FOREACH s_comp_index IN {2 .. 4}:
state.SewC[s_comp_index] =

IF (state.V == 0)
THEN 0
ELSE IF (state.V > parameters.Vmax/8)
THEN (1-parameters.sedimentation) * state.SewM[s_comp_index] / state.V
ELSE state.SewM[s_comp_index] / state.V;};


{FOREACH s_comp_index IN {5 .. NrOfSewComponents}:
state.SewC[s_comp_index] = IF (state.V == 0)
THEN 0
ELSE state.SewM[s_comp_index] / state.V ;};
```

FIGURE 31 EXTRACT OF THE STORMWATER TANK MODEL WHICH IS IMPLEMENTED IN THE BLEESBRUCK SEWER TRANSPORT MODEL (SOLVI, 2007)

As depicted in Figure 31 the sedimentation fraction (highlighted in yellow) is defined as 25 percent of the particulate matter within the basin. The green highlighted part of Figure 31 reveals that sedimentation takes place only if the wastewater volume is larger than one eighth of the tank capacity and according to this the outflowing particulate compounds are determined.

5.5.3 DYNAMICS AND CHARACTERISTICS OF THE BLEESBRUCK SEWER MODEL

In the following the dynamics of the Bleesbruck sewer system will be analysed. The aim is to understand the sewer behaviour simulated with the above model. In doing so, the diurnal fluctuations of wastewater pollutants during dry weather flow as well as the characteristics during rain events will be investigated. Furthermore, this preliminary analysis of the model built by Solvi (2007), is needed to compare model changes that will be carried out later within this work.

The simulation conducted with WEST® represents the wastewater flow rate and pollutant fluctuation of the Bleesbruck sewer system. The results depicted in Figure 32 reproduce sewer components from the 1st until the 16th of February. A relatively short time period was chosen for illustration in order to have a better view (resolution) on the sewer behaviour. The presented simulation results in Figure 32 stand for a reference condition that is to be considered when changes in the sewer model will be performed in the next paragraph.

Figure 32 shows that the pollutants and wastewater follow a certain diurnal pattern (as it was already described) during dry weather flow. The pollutants and wastewater show different diurnal fluctuations since a wastewater flow pattern and a pollutant pattern were independently implemented into the model structure. Moreover, all graphs show relatively low values during weekends (1st and 2nd, 8th and 9th, 15th and 16th day at Figure 32). This can be explained with a considerable influence of industrial pollution during the working days. The relatively small catchment exhibits almost 30% of the PE (population equivalent) caused by industry (brewery, dairy, slaughterhouse, roost-site and waste disposal) of the whole wastewater production. Thus, a notable weekend effect can be recognized during which less wastewater is discharged. During dry weather conditions are observed three significant peaks due to the flow as well as the pollutant pattern characteristic that are implemented in the sewer model. At peak hours (morning, noon and evening) more water is used by the population. Thus, higher flow rates and pollutant concentrations are expected at the treatment plant. Furthermore, it is to be mentioned that the sewer model does not distinguish between particulate and soluble pollutants variation during the course of the day when DWF is prevailing since the same fluctuation pattern (developed by SIDEN) is used in the model construction.

Sedimentation and accumulation effects are implemented in the sewer pipes as well as in stormwater tanks. This accumulated material of particulate pollutants is released during rain events. The first flush effect can easily be found when the amplitudes of the particulate components reflect the very significant peaks of the wastewater flow which reveal the rain events (e.g. during the 5th day in Figure 32). This erosion of accumulated material is dependent on the flow rate i.e. on the rain intensity as well as on the length of the period of time before when dry weather flow was prevailing and the particulate parts of the wastewater compounds could settle and accumulate. Yet, the first flush effect has no relevance for soluble pollutants whose load follows the same pattern which due to the additional rain water in the system leads to a lower concentration. The dilution effect is also to be expected for particulate material after the first flush.

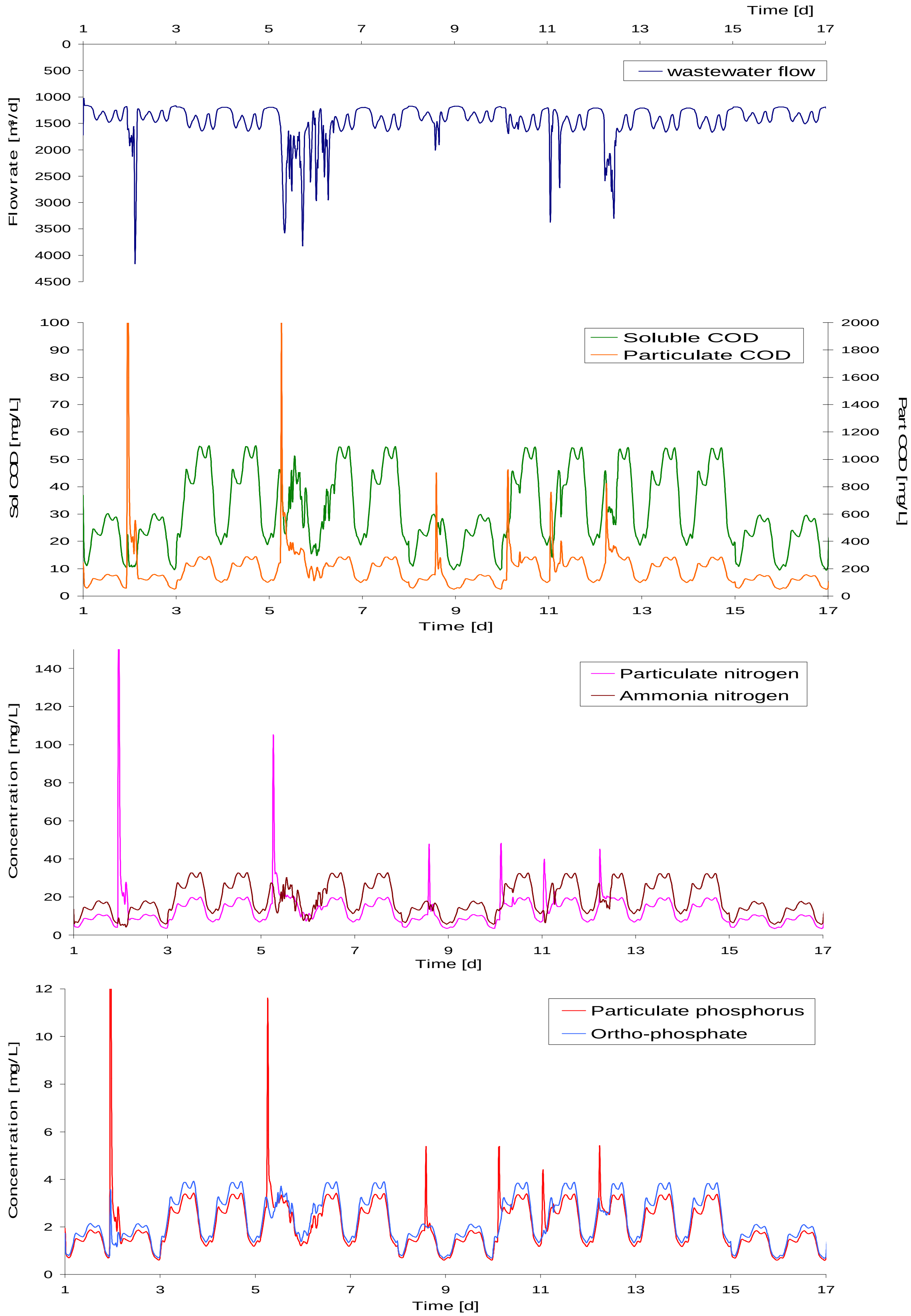


FIGURE 32 BLEESBRUCK SEWER SIMULATION (01. FEBRUARY – 16. FEBRUARY) DEPICTED ARE THE RESULTS FROM THE EFFLUENT OF THE SEWER SYSTEM (WWTP INLET)

5.5.4 MODIFICATION OF THE BLEESBRUCK SEWER MODEL

In chapter 5.5.1 - 5.5.3 the Bleesbruck sewer model was explained and analysed by simulation results. Furthermore, it was shown that only physical processes are implemented in the Bleesbruck sewer model. The load of the soluble and particulate organic matter during the course of the day is defined by the same hourly pattern within the work of Solvi (2007). In reality however, the load of biodegradable substrate fractions is subject to certain influences like industrial wastewater discharge, domestic water use and biological degradation processes taking place in the sewer system. These would lead to diurnal fluctuations in ratio to the total organic matter. In a nutshell, it can be expected that both fractions (soluble and particulate COD) show a different diurnal fluctuation. More particularly, hydrolysis is assumed to occur in the sewer system which causes changes in the ratio between readily and slowly biodegradable substrate. Within this work such diurnal fluctuations in the ratio to the total COD were checked by laboratory analysis for readily biodegradable substrate.

Regarding wastewater treatment, it can be expected that diurnal fluctuations of readily and slowly biodegradable substrate have a notable influence on the wastewater quality at the inlet of the treatment plant, hence, those variations are assumed to impact the treatment performance. A change to more slowly biodegradable substrate load in ratio to the total COD could provoke insufficient organic carbon elimination. More importantly, it could affect biological phosphorus elimination and denitrification processes in case they are implemented in the treatment plant since these processes are dependent on adequate readily biodegradable substrate supply.

This chapter describes simple sewer model changes that will be carried out. Later, it will be investigated whether these interventions can be observed in the output of the sewer model and how they influence the wastewater quality.

First, the effect will be analysed for the case an alteration is made to the readily and slowly biodegradable substrate ratio by changing the accumulating part of the wastewater components in the stormwater tank of the Bleesbruck sewer model. With this modification it will be verified whether sedimentation processes in the sewer system can be a reason for fluctuations in the particulate sewage components. For this purpose, the accumulation part in the stormwater tank implemented in the sewer model will be changed from 25% to 35%.

Secondly, a modification based to the hourly flow pattern that simulates the diurnal fluctuation will be carried out. As already pointed out the Bleesbruck sewer model takes only into account sedimentation and wash-off, respectively first flush effects which occur during the wastewater transport. But it is known that also biological processes like hydrolysis impact the soluble / particulate ratio of the wastewater components. In the sewer system, bacteria release enzymes that break down the particulate components and convert them into soluble fractions. To implement the hydrolysis process into the Bleesbruck sewer model a simple idea was used to describe this process mathematically.

One assumes that there is a significant relationship between the retention time of the wastewater in the sewer system and the extent of hydrolysis. The retention time is linked to the wastewater flow rate that fluctuates during the course of the day (DWF pattern). Thus, one can conclude that if the wastewater flow rate is relatively low the sewer retention time is longer and the more time the microorganisms have for hydrolysis (and vice versa). This relationship can be coupled to the hourly flow pattern implemented in the sewer model that describes the diurnal fluctuation of the Bleesbruck wastewater flow rate. The fluctuation is obtained by multiplying the mean flow with hourly factors between 0.3 and 1.9 depending on the daytime.

An exponential function (see equation (15)) has been developed to determine the part of the particulate compounds that is not being hydrolysed (f_{part}). This function shows a dependency on the flow factor f_i. ("state.FlowHelp" in Figure 34) which includes all DWF influencing parameters like infiltration, weekend effect, tourism effect and the hourly factor that is mainly responsible for the diurnal flow fluctuation. The latter is highly interesting for this investigation since the hydrolysis equation that will be implemented is dependent on the hourly factor. To calculate the hydrolysis dependency by using only this coefficient the time depending flow factor f_i is divided by a reference flow factor f_0 ("parameters.FlowHelp0" in Figure 34). Thus, the hourly flow factor that calculates the diurnal DWF fluctuation results and can be used further. Additionally, one assumes that if the retention time factor x_{rt} is equal or larger than a certain hourly factor no hydrolysis occurs since the retention time is too short. For the Bleesbruck sewer system a retention time factor of 2 was taken. Thus, in case of the Bleesbruck system hydrolysis occurs all over the day since the maximal hourly factor of the flow pattern is 1.9.

$$f_{part} = e^{\frac{f_i / f_0 - x_{rt}}{x_{rt}}} \qquad (if \;\; f_{part} > 1 \;\; then \;\; f_{part} = 1) \tag{15}$$

The exponential behaviour of equation (15) describing the influence of hydrolysis in the Bleesbruck sewer system is depicted in Figure 33.

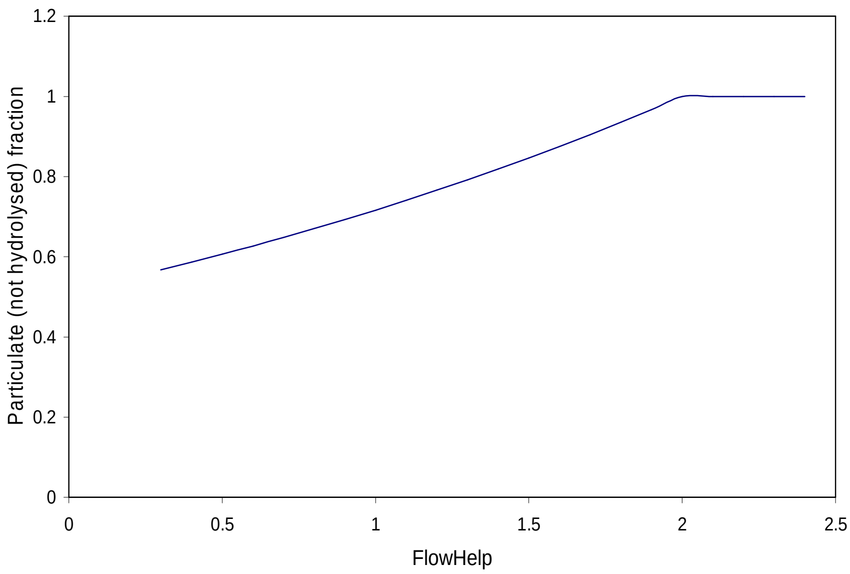


FIGURE 33 EXPONENTIAL DEPENDENCY OF HYDROLYSIS FRACTION ON FLOW RATE WHICH DESCRIBES THE WASTEWATER COMPOSITION FLUCTUATION IMPLEMENTED IN THE BLEESBRUCK SEWER MODEL (USING EQUATION (15) IF X_{RT} = 2)

Importantly, the model has to maintain the COD mass balance. Thus, the hydrolysed fraction 1-f_{part} of the particulate component is added to the soluble pollutant.

Summarizing, the hydrolysis process is implemented into the DWF generator of the sewer model by using the proposed function that is dependent on the diurnal fluctuation of wastewater flow. The hydrolysis function is multiplied with each particulate component (whereas the hydrolysed material is added to the corresponding soluble pollutants) that are furthermore multiplied by the hourly factors coming from the pollutant pattern to represent fluctuations (caused by different diurnal discharges) (See Figure 34).

```
{FOREACH s_comp_index IN {2 .. 4}:
      interface.Outflow[s_comp_index] = - parameters.Mean_S[s_comp_index] *
      state.PollutionHelp*
      FMin(1.0,exp((state.FlowHelp/parameters.FlowHelp0 - 2.0)/2.0));};

{FOREACH s_comp_index IN {5 .. NrOfSewComponents}:
      interface.Outflow[s_comp_index] = - parameters.Mean_S[s_comp_index] *
      state.PollutionHelp - parameters.Mean_S[s_comp_index-3] * state.PollutionHelp
      * (1.0-FMin(1.0,exp((state.FlowHelp/parameters.FlowHelp0 -2.0)/2.0)));};
```

FIGURE 34 IMPLEMENTATION OF HYDROLYSIS IN THE BLEESBRUCK SEWER MODEL. THE SEWER COMPONENT VECTOR DECLARES FROM INDEX 2 TO 4 THE PARTICULATE POLLUTANTS AND FROM INDEX 5 THE SOLUBLE POLLUTANTS

The conducted changes in the sewer model will be analysed in the following by simulations with WEST® and comparing the wastewater quality of the WWTP inlet with the one obtained for reference case as described above.

5.6 Results and discussion

5.6.1 Change of the sedimentation fraction in stormwater tanks

The simulation results obtained when the sedimentation fraction of the particulate material (in the stormwater tanks) was increased from 25% to 35% are depicted in Figure 35. The simulation results are presented from the 13th to the 19th of February and show the fluctuation at the inlet of the WWTP. This time span was chosen because the analysis of heavy rain events and dry weather flow is of special interest to reveal the expected differences in characteristics of the changed sewer model. On the 18th day a heavy rain event is registered with a peak flow rate of about 6300 m^3/d. The days before and the day after the 18th February dry weather flow characteristics can be observed. As can easily be recognized in this DWF period the values with 35% sedimentation show no difference compared to the simulation results with 25% sedimentation of the particulate material. Apparently, only at the end of the stormwater flow on the 18th day a difference between the two simulations can be found for a short while (marked with a green circle in Figure 35). The maximum difference amounts to approx. 46 mg/L particulate COD when the 35% model configuration is compared to the reference case.

If one compares the simulated values directly in the output file it was observed that less particulate material is arriving at the WWTP inlet during the rain event on the 18th day when the stormwater tank is configured with 35% sedimentation fraction. Often, the difference is really small and can hardly be recognized in a figure (see Appendix D). This phenomenon occurs until the night when the particulate material of the 35%-configuration starts to show significantly larger values for a short time.

These observations can be explained by the construction, respectively configuration of the stormwater tanks. The parallel construction of those tanks makes that only excess water enters the basin of the tank and is emptied when the flow capacities to the WWTP allow for it. Thus, the sedimented particles will only be flushed from the tank at the end of a rain event when the volume of water goes below a certain threshold, thereby representing flushing gates. Thus, sedimentation in the stormwater tank of the Bleesbruck sewer model takes only place during rain events when the water level in the basin reaches a certain value. Hence, no difference between the two simulations can be observed when dry weather is prevailing. After the rain event, when the water volume in the tank decreases and goes below the declared threshold, a cleaning flush (of an assumed fully mixed tank) is released. The cleaning flush explains the green encircled peak of particulates at the end of the 18th day. Furthermore, it has to be pointed out that only two stormwater tanks where sedimentation is taken into account are implemented in the Bleesbruck sewer model. The two tanks only handle around 5.15% of the DWF of the whole Bleesbruck catchment. Hence, the implemented 35% sedimentation of particulates in the tank does not result in a considerable change.

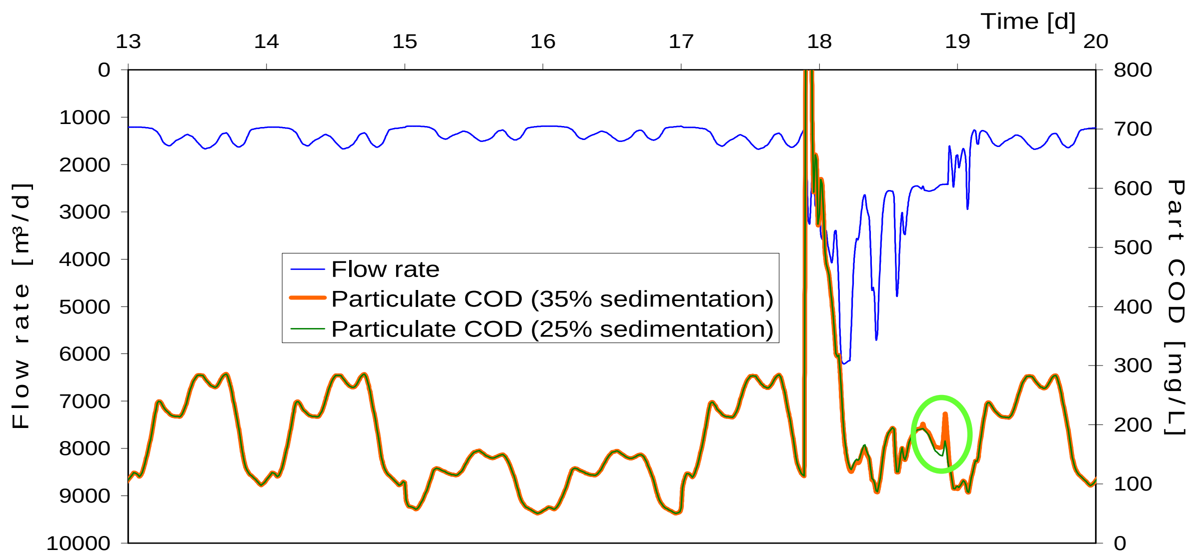


FIGURE 35 BLEESBRUCK SEWER MODEL SIMULATION. PRESENTED ARE THE RESULTS WHEN 25% AND 35% OF THE PARTICULATE MATERIAL SETTLE IN THE STORMWATER TANK.

5.6.2 IMPLEMENTATION OF HYDROLYSIS IN THE SEWER SYSTEM#

The described "hydrolysis equation" in chapter 5.5.4 was implemented in the Bleesbruck sewer model and the simulation results of particulate and soluble COD concentration are depicted in Figure 36. This work focuses mainly on the COD concentrations, its presented fluctuation is also representative for nitrogen and phosphorus fractions since they were calculated by the same extension with the hydrolysis model. The time period that was chosen to present the simulation results lasts from the 12^{th} to the 16^{th} of February. This time span is shown because both the characteristics of the dry weather flow pattern as well as the properties during rain events were to be presented. However, the rain event should not be too intense since a suitable resolution is needed for the illustration. The DWF characteristics can be observed from the 13^{th} to the 16^{th} day whereas the last two days in Figure 36 show the weekend DWF pattern.

In general it can be observed that, as can be expected, the particulate COD has a lower concentration if hydrolysis takes place. The part of particulate COD that is being hydrolysed can be found as an inverse in the soluble COD concentration. Furthermore, it stands out that the peak hour concentrations (during the morning, noon and evening) for particulate COD are less significant if hydrolysis is implemented in the model. Thus, almost 19% less particulate COD is found during diurnal flow peaks and almost 34% less concentration is calculated when the flow rate reaches its minimum during the night.

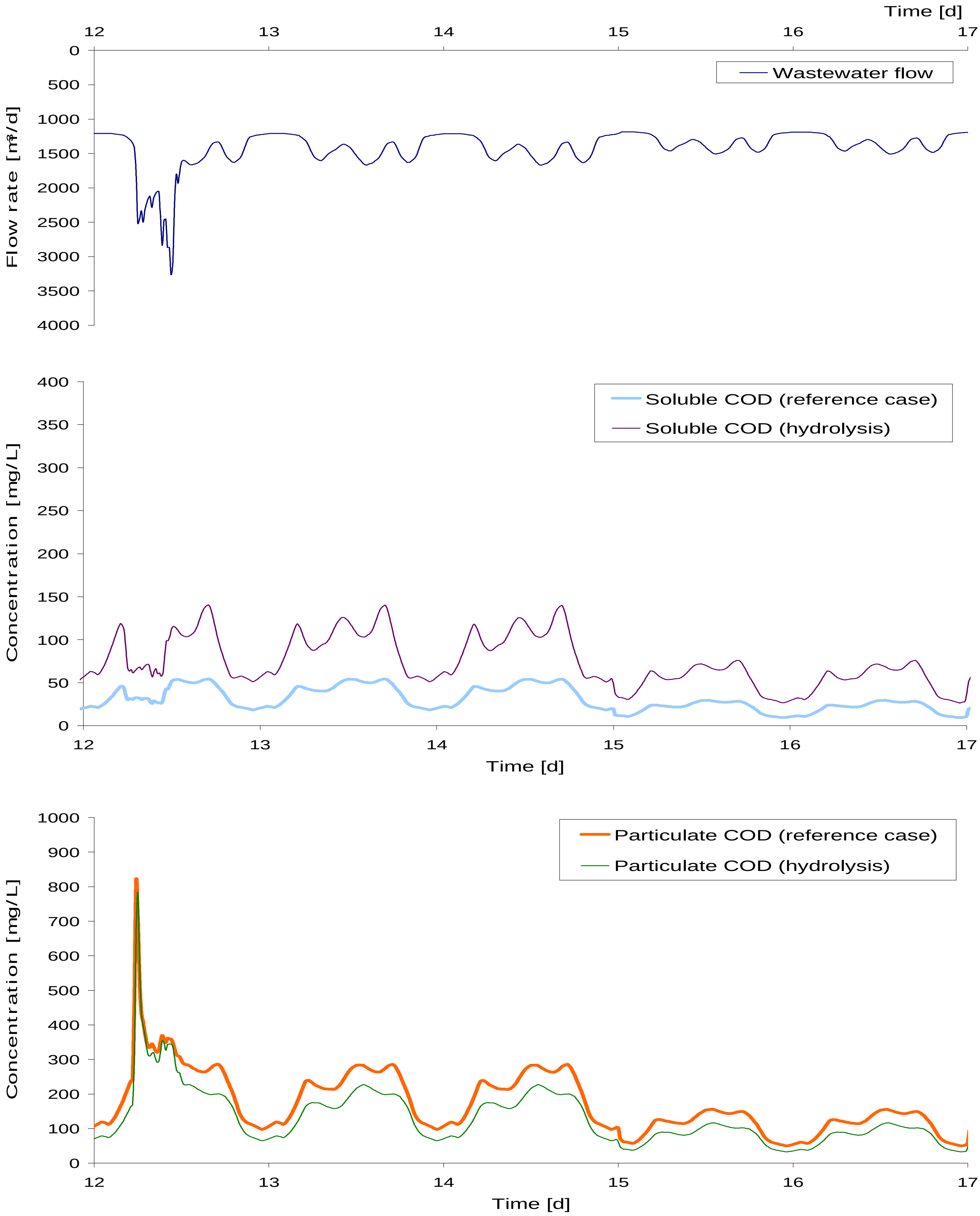


FIGURE 36 SIMULATION RESULTS OF THE BLEESBRUCK SEWER SYSTEM FROM THE 12TH TO THE 16TH OF FEBRUARY. THE DISTRIBUTION OF PARTICULATE COD AND SOLUBLE COD WHEN HYDROLYSIS IS IMPLEMENTED IS COMPARED TO THE REFERENCE CASE.

Huisman (2001) found out that biomass growth is limited during the night when the supply of nutrients is much smaller. He developed a hydrodynamic and biological sewer model which is based on the ASM No. 3 (Gujer et al. (1999)) and can be described as a WWTP model with biofilm properties in a sewer situation. Therefore, Huisman (2001) investigated biofilm development and its effect on wastewater pollutants in sewer systems. He pointed out that biofilms contribute considerably to conversion processes like hydrolysis and are responsible for two thirds of the aerobic activity during the night. Furthermore, he explains that the biofilm distribution is related to the hydrodynamic conditions of the sewer system (like reaeration rate) and the potential aerobic activity on the sewer wall is proportional to the TVS (total volatile solids).

However, within this work it is assumed that hydrolysis processes account for more influence on the wastewater quality when less wastewater is drained since this leads to a larger retention time as well as a larger ratio of moistened area / wastewater volume. In Figure 37 the diurnal distribution of the ratio hydrolysed particulate COD / total particulate COD (reference case) is shown under DWF conditions and reveals the fulfilment that was being pursued by implementing the hydrolysis process in the sewer model. The curve accounts for a smaller hydrolysed part when the flow rate is higher and a larger hydrolysis influence if less wastewater is discharged. Note that the two depicted distributions in Figure 37 are not proportional to each other since an exponential function was used to represent the hydrolysis process which is dependent on the diurnal flow pattern. Furthermore, the implemented hydrolysis process causes more soluble COD which is related to the reduced particulate material. The diurnal peaks of the soluble COD concentration are more significant when hydrolysis is taken into account. This phenomenon can be simply explained since 19% hydrolysis of a peak particulate COD concentration results in more soluble COD than 34% of particulate COD when low concentrations are determined in the sewer system. However, 19% hydrolysed material of the particulate COD is a relatively large value when only little hydrolysis impact is assumed during diurnal flow peaks. By altering the retention time factor x_{rt} implemented in the hydrolysis model (the simulation was carried out with $x_{rt} = 2$) the intensity of in-sewer hydrolysis can be adjusted to fit individual characteristics of the sewer systems. All in all it can be concluded that during dry weather flow around 76 kg particulate COD is being hydrolysed and accounts for the same amount of additional soluble COD. The implemented “hydrolysis equation” is related to the diurnal dry weather flow pattern and, thus, it also results in less particulate COD hydrolysis when additional rain water is in the sewer system as shown on the 12th day in Figure 36. This fact corresponds to the assumption that hydrolysis is negligible during storm events due to the short retention times in the sewer system. Here, the influence is very low if barely 4% less particulate COD concentration is calculated (due to hydrolysis) during the given rain event while the wastewater flow is 2560 m^3/d and the particulate COD concentration peak measures 780 mg/L.

Huisman (2001) describes that high shear forces of wastewater due to rain events result in erosion of the first (aerobic) biofilm layer from the sewer wall. The major part of this layer consists of heterotrophic bacteria, entrapped particles and bacteria storage material in the order of 5% of the TVS. Thus, a rain event would lead to a reduction in heterotrophic bacteria in the sewer system until the biofilm reaches its full structure again. Heterotrophic bacteria are responsible for hydrolysis processes so that a rain event influences hydrolysis not only due to lower retention time. This effect of biofilm erosion is not taken into account in the implemented hydrolysis model. As shown in Figure 36 the distribution of soluble and particulate COD reveals no impact on the hydrolysis potential after the rain event.

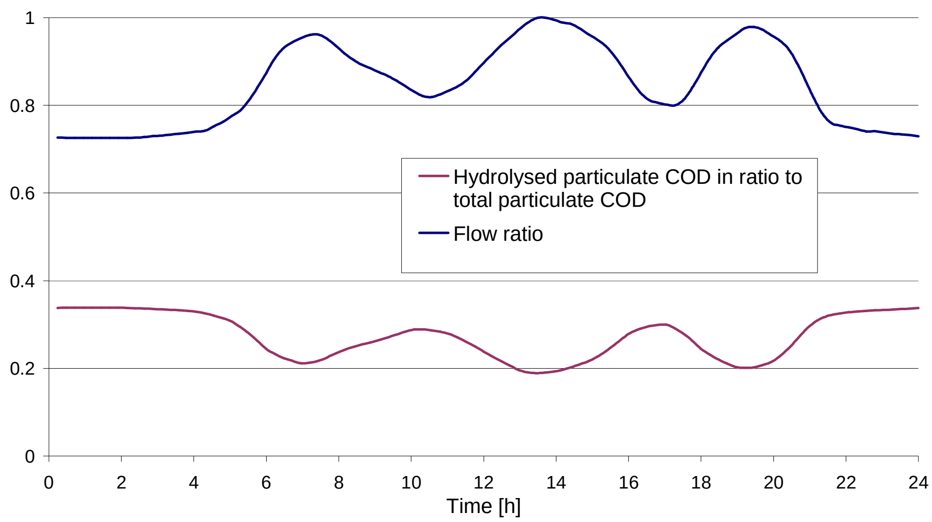


FIGURE 37 BLEESBRUCK SEWER MODEL (DWF) SIMULATION OF THE HYDROLYSED PARTICULATE COD PART DURING ONE DAY

When analysing the absolute values in Figure 36 one can generally deduce that the implemented hydrolysis still leads to more hydrolysed material during the day due to the pollutant concentration that is much higher than in the night (even if the hydrolysed fraction is much smaller as depicted in Figure 37). Thus, a maximum of 86 mg/L hydrolysed particulates is observed during the highest DWF peak of the day whereas only about 34 mg/L of hydrolysed particulate COD is determined during night hours, accounting for the according amount of additional soluble COD.

How the soluble load of the considered pollutants as fraction of the total amount of COD is changing at the inlet of the WWTP when hydrolysis is taken into account is depicted in Figure 38. One dry weather day is considered and the reference case is compared to the results of the sewer simulation taking into account hydrolysis processes. Furthermore, a distinction is made between working and weekend day since the hydrolysis implementation is dependent on the flow rate. The soluble COD fraction increases from 16% to 36% when hydrolysis is implemented in the sewer model and a working day is considered. Thus, more than the double diurnal load of soluble COD is expected to be treated. This large difference results from the very high mean concentration of particulate COD determined by Solvi (2007). For the other considered pollutants the ratios increase from 62% to 72% for the ammonia nitrogen fraction and from 53% to 65% for the ortho-phosphate fraction. However, the difference between working days and weekends is relatively small. Due to the mass balance that is fulfilled the inverse results of Figure 38 hold for the particulate / total pollutant load ratio.

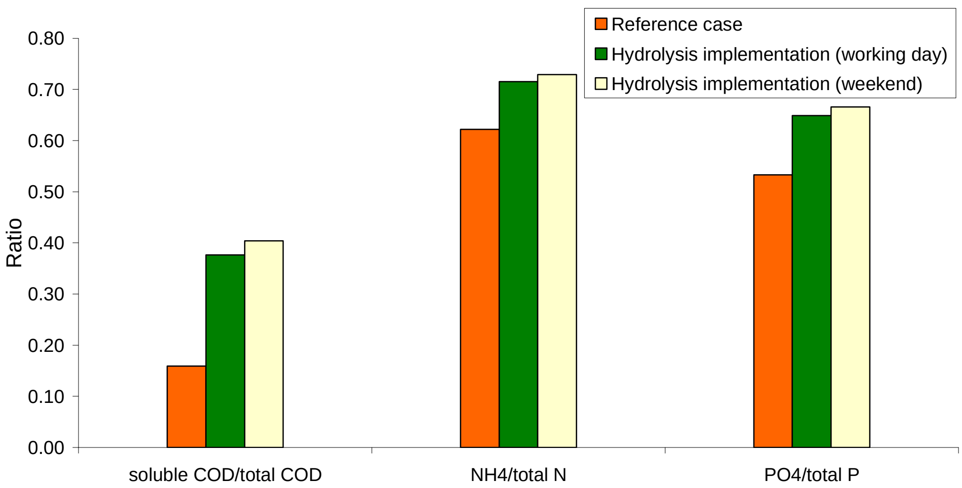


FIGURE 38 DIURNAL SOLUBLE LOADS AS FRACTION OF THE TOTAL AMOUNT OF EACH POLLUTANT THAT IS EXPECTED TO ARRIVE AT THE INLET OF THE BLEESBRUCK WWTP DURING DWF.

5.7 Conclusion

The first modification of the sewer model consisted of a change from 25 % to 35 % of the particulate fraction that is being settled in the stormwater tanks. The expected lower values during dry weather flow and larger magnitudes of first flush peaks could not be verified due to the tank, respectively model construction. The model only allows sedimentation when a certain water level in the tank is reached and releases the retained wastewater when the flow capacities to the WWTP allow for it. A very considerable rain event was chosen for the evaluation but the resulting cleaning flush effect of the stormwater tanks is relatively low since the dilution of rain water has to be taken into account as well. Additionally, it must be considered that only two stormwater tanks which allow sedimentation processes are implemented in the Bleesbruck sewer model.

Thus, while it could be shown that sedimentation processes in stormwater tanks can be a reason for particulate COD fluctuation at the inlet of the wastewater treatment plant, the observed influences on the general fluctuation are negligible considering the Bleesbruck system with few stormwater tanks where sedimentation takes place.

The second modification applied to the Bleesbruck sewer model was the implementation of hydrolysis processes. It is to be pointed out that the implemented hydrolysis model meets the idea that was being pursued. Regarding the wastewater composition a conversion from particulate to soluble material dependent on the wastewater flow rate could be achieved. Longer retention times and lower wastewater flow rates would stimulate hydrolysis even if the biological wastewater activity is inversely dependent on the flow rate according to Huisman (2001). Heterotrophic bacteria are located in the wastewater and especially at solid / liquid (sewer wall / wastewater) interface where biofilm development constitutes a major part of the biological processes e.g. hydrolysis where particulate wastewater pollutants (particulate COD, particulate nitrogen and particulate phosphorus) are converted into soluble material that is directly available for microorganisms. By using the implemented hydrolysis model it was shown that 34% of the particulate COD is broken down (by the released enzymes) into the proportionate amount of soluble substrate when the flow rate reaches its minimum during the night. During maximal dry weather flow the hydrolysed fraction of the particulate COD reduces to 19%.

If the diurnal (dry weather) load of the considered wastewater contents is taken into account and compared the results reveal a very large hydrolysis effect on the organic content with 22% of the total diurnal COD load being converted. In contrast a 10% conversion of the total nitrogen load from particulate nitrogen to ammonia nitrogen was observed while a similar conversion process of 12 % was determined for the phosphorus load.

In summary, it could be shown that the implemented hydrolysis model gives reasonable results with regard to the way how the hydrolysis process could impact the particulate and soluble pollutant fluctuations during the course of the day. The model will be especially suitable in the field of integrated modelling where simplicity is a considerable advantage in terms of transparency and computing time. By altering the retention time factor the model can be calibrated to fit the intensity of hydrolysis of an individual sewer system. Regarding the organic wastewater components it was shown that hydrolysis has, in contrast to biodegradation and sedimentation / resuspension processes, no influence on diurnal fluctuations of the total organic material in the sewer system. It rather affects the particulate and soluble ratio of the total COD. Soluble COD consists of inert soluble organic material S_I and readily biodegradable substrate (which is divided into fermentable, readily biodegradable organic substrate S_F and fermentation products S_A) to describe the biological processes at the Bleesbruck treatment plant according to ASM No.2d (Henze et al. (1999)). The particulate part of the COD is subdivided (according to a transfer factor at the interface between sewer system and WWTP) into inert particulate organic material X_I, heterotrophic organisms X_H and slowly biodegradable substrate X_S. The in-sewer hydrolysis process would therefore cause less slowly biodegradable substrate since it amounts to 80% of the particulate COD and more readily biodegradable substrate S_F and S_A which comprise 80% of the soluble COD at the Bleesbruck WWTP inlet. Thus, in-sewer hydrolysis is assumed to significantly influence the treatment performance with respect to design and operation that are closely linked to efficiency and costs of a wastewater treatment plant.

6 Final conclusion

6.1 Resume and limits of the thesis

To improve the integrated urban wastewater modelling with respect to its simulation accuracy the modelling approach takes more and more processes into account. The process simulation is dependent on the state of the art regarding knowledge about physical, chemical and biological processes and its measuring technology in all three sections belonging to the urban wastewater system: sewer system, wastewater treatment plant and river.

This work dealt with the fluctuation of readily and slowly biodegradable substrate concentrations in sewer systems during the course of the day. Respirometric analysis was carried out to investigate those COD fractions defined according to the ASM (Henze et al. (2000)). For that purpose, wastewater was sampled at three different times on the day at the inlet of the Saint Nicolas WWTP (Québec, Canada). A fluctuation of readily biodegradable substrate with a maximal fluctuation of 10% of the total COD was observed. In contrast, a significant variation of the slowly biodegradable substrate could not be found. Thus, in-sewer hydrolysis processes where particulate material is converted to soluble material do not seem likely so that biological degradation processes or changes in wastewater discharge must have caused the fluctuation of readily biodegradable material. It has to be mentioned that snowmelt during the sampling could have influenced the measurements significantly. Moreover, low temperatures and the relatively small size of the sewer system (in Saint Nicolas) impact the biological activity.

From a modelling point of view it was investigated how fluctuations of soluble and particulate organic material are considered in integrated sewer systems. For this a sewer model built and calibrated by Solvi (2007) to simulate an urban wastewater system in Luxembourg was analysed and used to investigate the effect of sewer processes on wastewater quality at the WWTP inlet. By the modelled sedimentation fraction of particulates in stormwater tanks it was shown that sedimentation processes in the sewer system have a considerable influence on the particulate pollutant concentration.

A hydrolysis process was implemented in the Bleesbruck sewer model taking into account that a part of the particulate wastewater pollutants (in terms of COD, nitrogen and phosphorus) are broken down to their soluble form by bacteria released enzymes. This process is assumed to be mainly dependent on the wastewater flow rate where an exponential relationship with the sewer retention time was proposed. Accordingly, this biochemical process was modelled. The intensity of the hydrolysis effect can be altered by the implemented retention time factor within the model. Thus, the user is able to adjust the hydrolysis process according to the individual sewer characteristics. Sewer simulations were conducted and it was shown that the hydrolysis implementation gives reasonable results regarding the hydrolysis influence on the diurnal flow pattern. A firm conclusion about the influence on wastewater quality at the inlet of the Bleesbruck WWTP could not be drawn since a calibration with field data was not conducted within this work. But the obtained simulation indicates that in-sewer hydrolysis processes can have considerable relevance for design -, operation - and costs of wastewater treatment plants considering that processes of denitrification and biological phosphorus removal depend on readily biodegradable substrate. However, for each individual case it has to be verified whether this process is to be taken into account since the biological activity responsible for hydrolysis in sewer systems is sensitive to temperature and wastewater retention time.

6.2 Perspectives

In future work a full-scale test series of respirometric analysis should be carried out to get more qualitative and detailed information about the readily and slowly biodegradable substrate concentration fluctuations during the course of the day. Furthermore, field measurements are to be considered to examine the dependencies of hydrolysis in sewer systems more closely. For example the influence of temperature or the assumption that the sewer system shows less biological activity after a rain event due to biofilm erosion was not investigated within this work. Moreover, a benchmark sewer system and wastewater treatment plant with a relative simple and clear structure is recommended to analyse the impacts of in-sewer hydrolysis on the treatment performance in detail so that general qualitative assumptions can be derived and further used for other wastewater systems.

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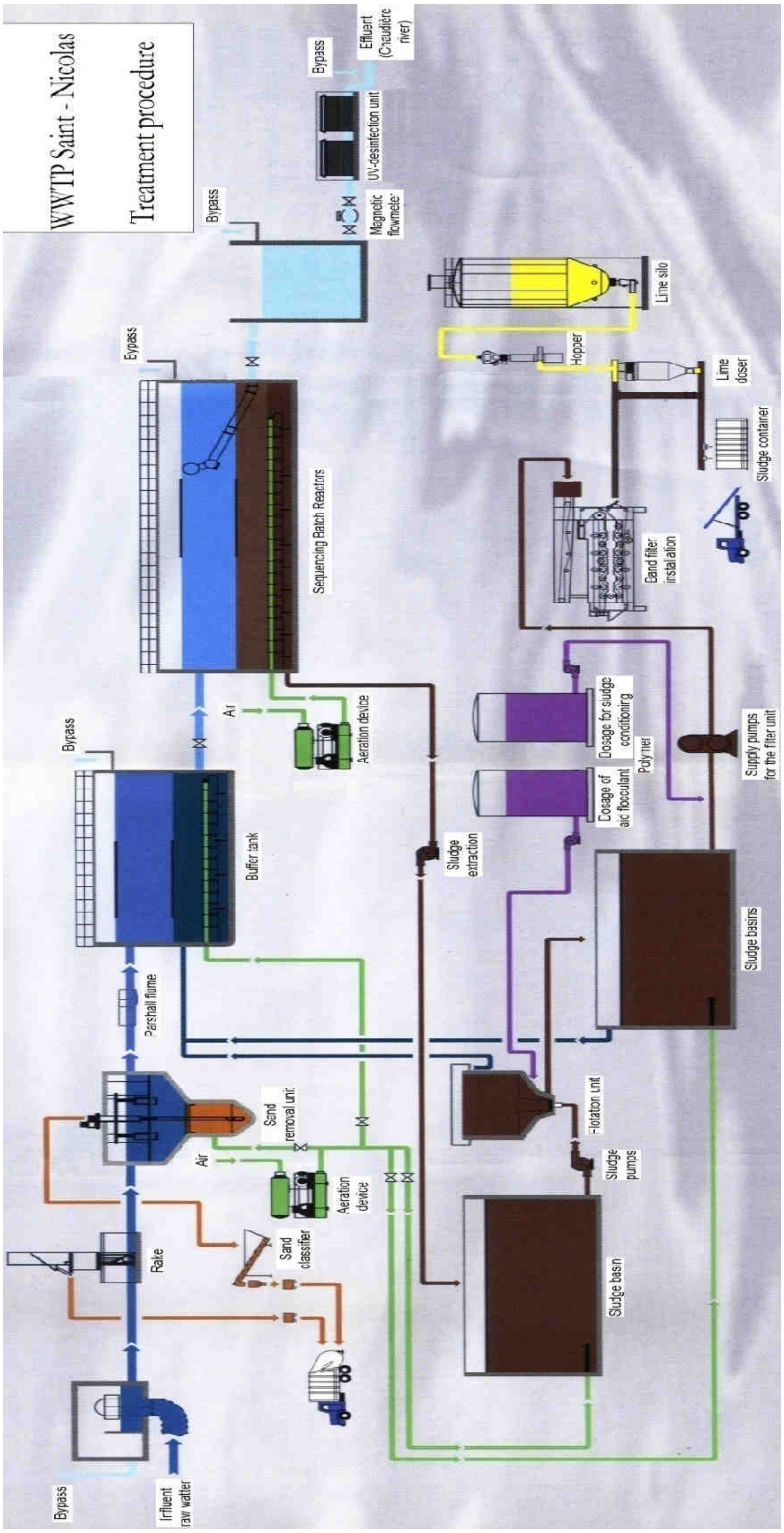
8 APPENDICES

APPENDIX A

Process chart of the Saint - Nicolas WWTP



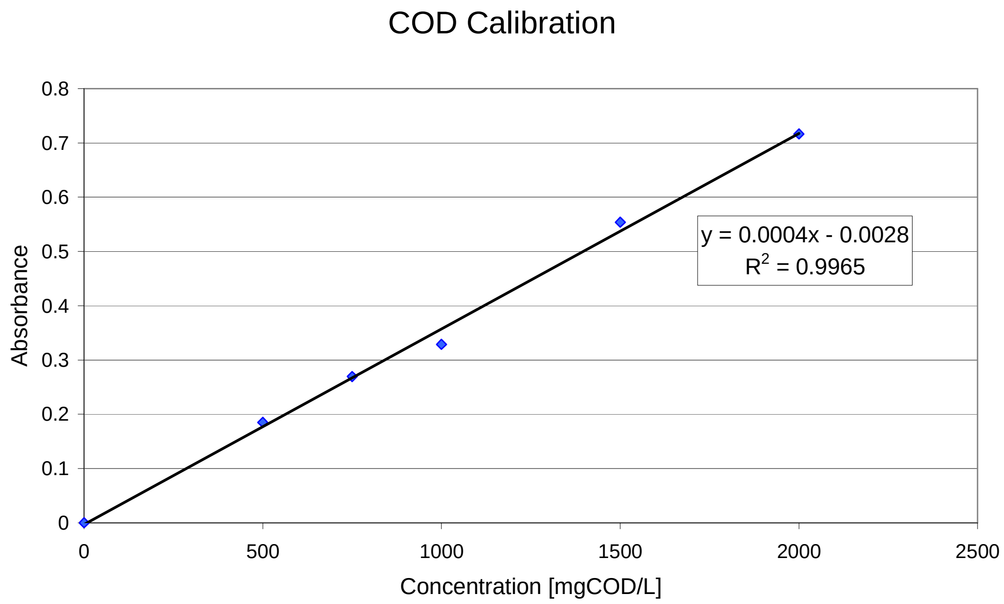
APPENDIX B

Confidence level [%] dependent on the degrees of freedom → Used for the Nalimov outlier tests

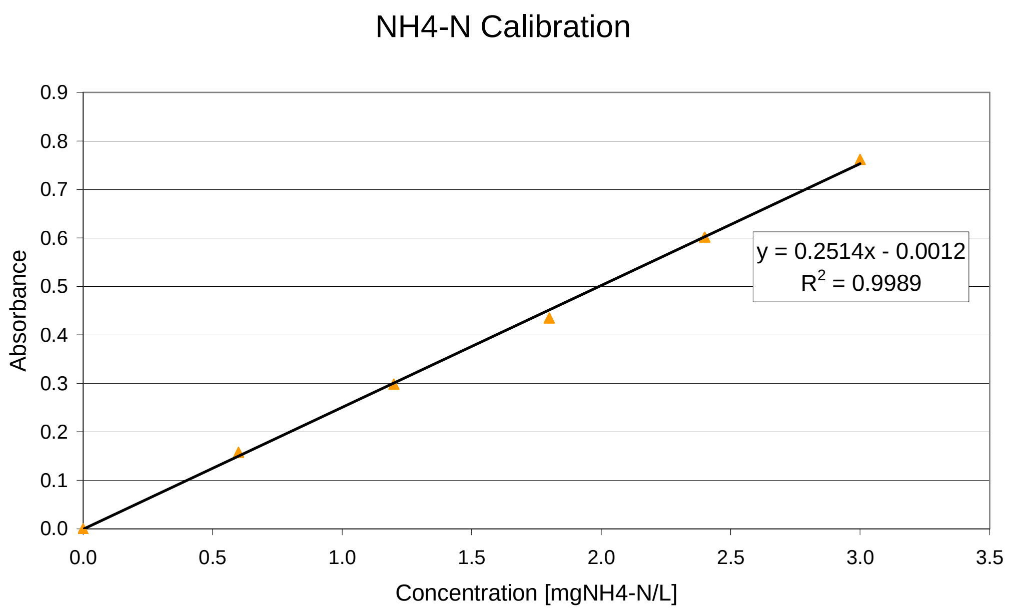
Degrees of freedom	95 %	99 %	99,9 %
1	1,409	1,414	1,414
2	1,645	1,715	1,730
3	1,757	1,918	1,982
4	1,814	2,051	2,178
5	1,848	2,142	2,329
6	1,870	2,208	2,447
7	1,885	2,256	2,540
8	1,895	2,294	2,616
9	1,903	2,324	2,678
10	1,910	2,348	2,730
12	1,920	2,385	2,812
14	1,926	2,412	2,874
16	1,931	2,432	2,921
18	1,935	2,447	2,959
20	1,937	2,460	2,990
25	1,942	2,483	3,047
30	1,945	2,498	3,085
40	1,949	2,518	3,134
50	1,951	2,529	3,166
100	1,956	2,553	3,227
200	1,958	2,564	3,265
500	1,959	2,570	3,279
∞	1,960	2,576	3,291

APPENDIX C

Calibration of the spectrophotometer for COD measurements



Calibration of the spectrophotometer for NH_4^+-N measurements



APPENDIX D

Extract of the Bleesbruck sewer simulation – The results of 25% and 35% sedimentation fraction of particulates in the stormwater tanks are compared during a rain weather event.