MODELING ACTIVATED SLUDGE POPULATION DYNAMICS

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ABSTRACT

MODELING ACTIVATED SLUDGE POPULATION DYNAMICS

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Population dynamics play an important role in the performance of biological treatment by activated sludge. Current models often do not take this into consideration with concomitant failure in prediction of the outcome. Therefore this study aims at incorporating kinetics of population dynamics into ASM1 Activated Sludge Model. The AEROFIL model was used in the study for modeling population dynamics. The literature data was used for verification and model supporting. The already existing AEROFIL model was developed by using the AQUASIM utility program.

Basic requirement of treatment by activated sludge is the floc

formation. Any failure in activated sludge floc formation results in decrease in treatment efficiency.

The AEROFIL model used in this study is capable of predicting the behavior facultative aerobic floc-forming and obligate aerobic filamentous bacteria as well as nitrifying microorganisms in the case of aerobic bulking. This way it is now possible to predict activated sludge biocenoses, hence sludge settleability.

Effects of high loadings with readily biodegradable substrate, flow pattern in longish aeration basins, inoculation by facultative aerobic floc-former microorganisms, inoculation by obligate aerobic filamentous microorganisms on the activated sludge biocenoses is investigated. Besides, effects of aerobic, anaerobic and anoxic selectors and selector compartmentalization on the *filamentous microorganism to total biomass ratio* were analyzed by using the AEROFIL model in AQUASIM program. The predictions were compared with the literature. Moreover, population dynamics in some of the biological nutrient removing activated sludge modifications like A/O, A^2/O and Bardenpho were investigated and compared to a completely mixed activated sludge process.

Keywords: Activated sludge bulking, population dynamics, modeling, AEROFIL, AQUASIM

ÖZ

AKTİF ÇAMUR POPULASYON DİNAMİKLERİNİN MODELLENMESİ

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Aktif çamur türü biyolojik arıtım sistemlerinde populasyon dinamikleri çok önem taşımaktadır. Kullanımdaki modeller sıklıkla bu hususu göz ardı etmekte ve sonuçta yetersiz kestirimler elde edilmektedir. Bu çalışmada populasyon kinetiklerinin ASM1 modeline monte edilmesi amaçlanmaktadır. Araştırmada AEROFIL modeli ve literatür verileri kullanılarak modelin doğruluğu kanıtlanmıştır. Modelin bilgisayara uyarlanması AQUASIM genel amaçlı program kullanılarak yapılmıştır.

Atıksuların aktif çamur işlemiyle arıtılmasının temel prensibi flok oluşumudur. Aktif çamur flok oluşumunda meydana gelebilecek herhangi bir bozulma arıtım veriminin

düşmesiyle sonuçlanacaktır.

Bu çalışmada aktif çamur florasını meydana getiren fakültatif flok bakterileri, mecburi aerobik ipliksiler ve nitrifikasyon mikroorganizma konsantrasyonları kestirilmeye çalışılmıştır.

Çalışmada, yüksek miktarda biyolojik olarak kolay bozunabilen madde yüklemesi ve dar uzun geometrili havalandırmalı tanklarındaki hidrolik koşulların aktif çamur florası üzerindeki etkileri araştırılmıştır. Bunun yanı sıra aktif çamura fakültatif flok bakterisi ve mecburi aerobik ipliksi bakteri aşılanması, aerobik, anoksik ve anaerobik selektörlerin ve selektör kademelendirilmesinin *ipliksi mikroorganizma/toplam biyokütle* oranı üzerine etkileri AQUASIM programı içinde AEROFIL modeli kullanılarak kestirilmiştir. Çalışmadan elde edilen kestirimler literatür değerleri ile karşılaştırılmıştır. Ayrıca, A/O, A^2/O ve Bardenpho gibi çeşitli biyolojik nütriyent giderici ileri aktif çamur konfigurasyonlarının da flok yapısı üzerine etkileri tam karışımlı aktif çamur ile karşılaştırmalı olarak belirlenmiştir.

Anahtar Kelimeler: Aktif çamur şişmesi, popülasyon dinamikleri, modelleme, AEROFIL, AQUASIM

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LIST OF SYMBOLS AND ABBREVIATIONS

b_A	Decay coefficient for autotrophic biomass
b_H	Decay coefficient for heterotrophic biomass
f_P	Fraction of biomass leading to particulate products
i_{XB}	Mass of nitrogen per mass of COD in biomass
i_{XP}	Mass of nitrogen per mass COD in products from biomass
k_a	Ammonification rate
k_h	Maximum specific hydrolysis constant
K_{NH}	Ammonia half-saturation coefficient for autotrophic biomass
K_{NO}	Nitrate half-saturation coefficient for denitrifying heterotrophic biomass
$K_{O,A}$	Oxygen half-saturation coefficient for autotrophic biomass
$K_{O,H}$	Oxygen half-saturation coefficient for heterotrophic biomass
K_S	Half-saturation coefficient for readily biodegradable substrate
K_X	Half-saturation coefficient for hydrolysis of slowly biodegradable substrate
μ_A	Maximum specific growth rate of autotrophic biomass
μ_H	Maximum specific growth rate of heterotrophic biomass
η_g	Correction factor for μ_H under anoxic conditions

η_h	Correction factor for hydrolysis under anoxic conditions
Y_A	Autotrophic yield coefficient
Y_H	Heterotrophic yield coefficient

AEROFIL	Mathematical Model for Aerobic Bulking, AERObic FILamentous
ASIM	Activated Sludge SIMulation Program
AQUASIM	Computer Program for SIMulation of AQUatic Systems
A/O	Anaerobic / Oxic
A^2/O	Anaerobic / Anoxic / Oxic
ASM	Activated Sludge Model
BNR	Biological Nutrient Removal
BOD	Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
F/M	Food to Microorganism Ratio
IAWPRC	International Association on Water Pollution Research and Control
MGD	Million Gallons per Day
MLSS	Mixed Liquor Suspended Solids
SRT	Sludge Retention Time
VSS	Volatile Suspended Solids
$X_{Fil}/X_{Biomass}$	Filamentous Microorganism to Total Biomass Ratio

CHAPTER 1

INTRODUCTION

Biological treatment processes are used to convert the finely divided and dissolved organic matter in wastewater into flocculent settleable solids that can be removed in sedimentation tanks. Although these processes also called secondary treatment processes are employed in conjunction with physical and chemical processes, which are used for preliminary treatment, they are by no means substitutes. Primary sedimentation is most efficient in removing coarse solids, whereas biological processes are most efficient in removing organic substances that are soluble or in the colloidal-size range (Tchobanoglous and Burton, 1991).

The activated sludge process is the most popular aerobic method used for biologically treating wastewater (Biggs and Lant, 2002). The process may be defined as a method of sewage treatment in artificially aerated vessels. The organic constituents of sewage are metabolised by the action of activated sludge. As a result, organics are partially or completely degraded and/or mineralized. The microorganisms growing within the biological flocks are termed *activated sludge*

biomass, hence the process is accordingly known as the activated sludge process (Hanel, 1988).

The process depends on the principle of aerating the settled wastewater. Meanwhile, microorganisms utilize the dissolved and suspended organic materials in the wastewater in order to fulfill their metabolic requirements and to get energy for growth (Muslu, 1996).

Bacteria, which are essential for the process, remove the soluble and insoluble pollutants by using them as substrates for metabolism. Bacteria exist in the system as aggregates, which are termed flocs. Flocs are composed of heterogeneous flocculated masses of cells, along with organic and inorganic materials captured within the floc structure. This is called activated sludge (Biggs and Lant, 2002).

Activated sludge is comprised of three levels. The first level is made up of bacteria tightly bound together by a polymeric matrix to form the microcolonies. The microcolonies constitute the second level. Finally microcolonies aggregate to form the final activated sludge flocs (Biggs and Lant, 2002).

The formation of flocs is essential for the operation and efficiency of the treatment process (Wanner, 1994). If flocculation fails to occur, active biomass, which is essential for the operation of the process is lost from the system. Not only does this reduces the process efficiency but may result in excess of solids being discharged

into the environment. Population dynamics may be a suitable framework for modeling activated sludge flocculation.

1.1 Scope of the Study

The scope of this study is to incorporate population dynamics into the current activated sludge model AEROFIL and to develop this model with AQUASIM utility program. The study will also involve verification of the model and model supporting examples by using literature data and finally to investigate the population dynamics in several advanced activated sludge modifications by using the developed computer code.

CHAPTER 2

THEORETICAL BACKGROUND

2.1 Activated Sludge Process Fundamentals

Activated sludge process is a rather unique biotechnological process and it does not have many similarities to other processes that are frequently called "fermentation". Typical features of the activated sludge process are:

1. Multi substrates in terms of chemical composition and variety of particle sizes
2. Multi species biological culture, growing in aggregates called flocs
3. Fluctuating flows, temperatures, influent wastewater composition
4. Metabolizing, oxidizing, reducing, polymerizing compounds
5. Different reactor configurations like completely mixed tanks, plug-flow, anoxic, oxic, and anaerobic selectors, sequencing batch reactors.

The activated sludge process has a large number of modifications and variations. In the last decade, important developments took place in industrial wastewater treatment by activated sludge and nutrients removal (N and P) have been a major

issue in the design of activated sludge process. Moreover solids separation problems such as bulking, foaming, turbid effluents etc. have all been addressed in the recent decade (Eckenfelder and Grau, 1992).

2.2 Kinetic Model Development

With few exceptions, kinetic models which have been proposed to describe the activated sludge process have been developed on the basis of steady-state conditions. In the discussion to follow, commonly use of these kinetic models are presented and the assumptions made during development of each model are outlined. Such an approach is necessary if one is to understand the limitations of each model when used for plant design (Benefield and Randall, 1980).

A typical flow scheme for a completely mixed activated sludge process is shown in Figure 2.1. In this figure, Q represents the rate of raw wastewater flow to the aeration tank; S_0, the substrate concentration in the raw wastewater; V_a, volume of the aeration tank; X, biomass concentration in the aeration tank and in the effluent; S_e, represents the steady-state substrate concentration after treatment; Q_r, is the sludge recycle flow rate; R, is sludge recycle ratio (i.e., Q_r/Q); Q_W, is the rate of sludge wastage; and X_r; represents the biomass concentration in the underflow from the secondary clarifier (Benefield and Randall, 1980).

The sludge wastage can be accomplished either from sludge return stream or directly from the aeration tank, as shown in Figure 2.1. Although sludge wasting from the

sludge return line is traditional, sludge wasting from the aeration tank is more desirable as this method offers an opportunity for better plant control and is also beneficial in subsequent sludge thickening operations. It has been shown that higher solids concentrations can be achieved when dilute mixed liquor rather than concentrated return sludge is thickened. Model development presented below is based on sludge wastage from aeration tank (Benefield and Randall, 1980).

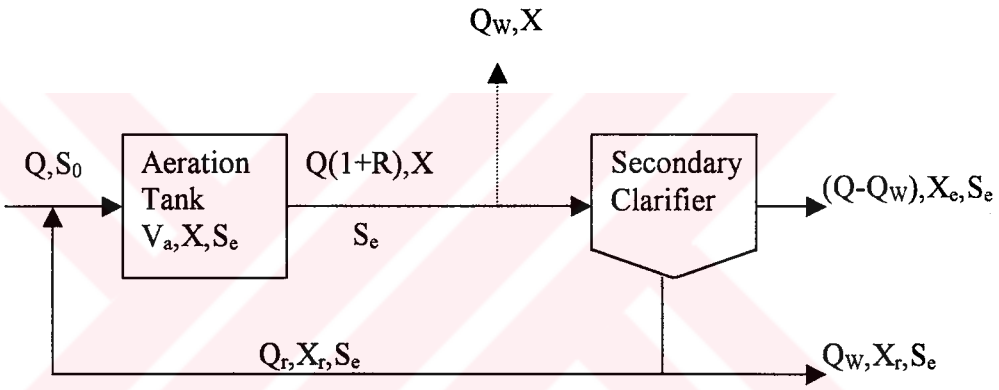

Figure 2.1 Typical flow scheme for a completely mixed activated sludge plant (Benefield and Randall, 1980)

Material-balance equations are written for scheme shown in Figure 2.1. Development of appropriate kinetic equations are based on this material-balance and on the following assumptions:

1. Complete mixing is achieved in the aeration tank.
2. Influent substrate concentration remains constant.

3. No microbial solids are present in the raw wastewater.
4. No microbial activity occurs in the secondary clarifier.
5. No sludge accumulates in the secondary clarifier and a reasonable efficiency of solids-liquid separation is accomplished.
6. All biodegradable substrate is in the soluble form.
7. Steady-state conditions prevail throughout the system.

The important operational parameter, called *biological solids retention time* (BSRT or sometimes just SRT), symbolized by θ_C, is defined as the average time a unit of biomass remains in the treatment system (Benefield and Randall, 1980).

$$\theta_C \equiv \frac{(X)_T}{(\Delta X/\Delta t)_T} \qquad 2.1$$

Where,

$(X)_T$ = total active biomass in treatment system, mass

$(\Delta X/\Delta t)_T$ = total quantity of active biomass withdrawn from the system daily, mass time^{-1}; this includes solids purposely wasted plus those lost in the effluent.

For a completely mixed activated sludge process with solids recycle, following the assumption previously outlined, equation 2.1 can be written as

$$\theta_C = \frac{XV_a}{Q_W X + (Q - Q_W)X_e} \qquad 2.2$$

Furthermore, for steady-state conditions,

$$\theta_C = \frac{1}{\mu} \qquad 2.3$$

Where, μ represents the rate of growth per unit amount of biomass and is termed *the specific growth rate*. Hence, the importance of θ_C as a control parameter is apparent.

By controlling θ_C one controls the specific growth rate and thus the physiological state of the organisms in the system.

The independence between effluent and influent substrate concentrations, as implied in activated sludge kinetic equations, is of great importance in the design and control of wastewater treatment plants. For example, when θ_C is used for plant control, there is no requirement to evaluate the mixed liquor volatile suspended solids (MLVSS), which is generally taken to be a measure of the biomass in the system, nor is there a need to monitor the influent and effluent substrate concentrations (measured as BOD_5, COD, or TOC). As long as θ_C is held constant, any change in influent substrate concentration will result only in a change in the steady-state biomass concentration while the effluent quality will remain constant. Because of this, it has been proposed that θ_C should be controlled by hydraulic means. Furthermore, because of this simplicity, the method of hydraulic control has become popular in actual plant operations.

Moreover independence between influent and effluent substrate concentrations also suggests that when determining the kinetic coefficients required for process design, it may not be necessary to use wastewater samples of identical concentrations as with the field conditions (Benefield and Randall, 1980).

2.3 Activated Sludge Modifications

Difficulties are encountered in the operation of activated sludge plants in the face of the complex and variable nature of the wastes involved. These difficulties triggered extensive efforts to review existing design criteria and operating practices. Therefore, process modifications were developed that would permit existing plants to treat larger flows and greater loads while maintaining a high effluent quality. While the problems were recognized, they were not always thoroughly and scientifically defined so that rational corrective measures could be taken. Therefore, experience and good judgement had to form the basis for plant improvements and modifications. Several original concepts and process modifications emerged from the individual works gained universal acceptance in sewage treatment practice (Orhon and Artan, 1994).

Conventional Activated Sludge Treatment

The objective of an activated sludge process is to remove soluble and insoluble organics from a wastewater stream and to convert this material into a flocculant microbial suspension. Classically, this was accomplished by mixing the wastewater with a biological culture in a long, narrow aeration basin with a volume sufficient to provide 6 to 8 h contact period between the two components. The biomass was then separated from the liquid stream in a secondary clarifier. A portion of this biological sludge was wasted and the remainder was returned to the head of aeration tank. Such a process arrangement has been termed *conventional activated sludge treatment*

(Benefield and Randall, 1980). Because of inherent deficiencies numerous modifications to the original conventional process have been proposed.

Tapered Aeration

Flow pattern of the tapered aeration modification is identical to the conventional process. The actual difference between the two processes is in the diffuser arrangement. In tapered aeration the diffusers are spaced so that more air is supplied at the head of the tank, where the oxygen demand is greatest, and decreases along the tank length as the demand decreases. Such an arrangement is more economical than supplying a constant amount of air along the entire tank (Benefield and Randall, 1980).

Step-Aeration

In this modification, return sludge is mixed with a portion of the wastewater and enters the head of the aeration tank. Wastewater is also fed into the tank at different points along its length. Advantages of this process modification are (1) better equalization of waste load, (2) lower peak oxygen demand, (3) better distribution of oxygen demand over tank length, and (4) smaller overall aeration tank volume. This process is normally designed as a series of completely mixed reactors (Benefield and Randall, 1980).

High-Rate Activated Sludge

For this particular process modification a low mixed liquor suspended solids concentration is maintained in the aeration tank. Under such a condition, the process yield is maximized, which implies that the total oxygen requirement per unit of substrate removed per unit time is minimized even though the specific oxygen utilization rate is high. Although the organic removal rate is high on a unit biomass basis, the overall removal is not. Because of this characteristic, the high-rate process cannot be used where a high quality effluent is required (Benefield and Randall, 1980).

Complete Mixing

Through the use of complete mixing it is possible to establish a constant oxygen demand as well as a uniform solids concentration throughout the tank volume. Furthermore, this process is highly resistant to upset from shock loadings because of the rapid blending of feed and tank contents. As a result, complete mixing has become very popular in the choice of mixing regimes (Benefield and Randall, 1980).

Extended Aeration

Extended aeration plants are generally small (applicable to flows less than 1MGD) because of the large aeration tank volumes required and they almost invariably employ complete mixing. Since there is no sludge wasting, food/microorganism

(F:M) ratio will be used as the loading criterion. Extended aeration process maximizes the total oxygen requirement per unit of substrate removed per unit time and thus the associated energy cost (Benefield and Randall, 1980).

Contact Stabilization

Two aeration tanks are provided. The first tank provides contact between the biomass and the wastewater. It operates on a short retention time, sufficient only for the transfer of substrate from the liquid to the solid phase. The biomass is then separated from the wastewater in a secondary clarifier and the biological sludge channeled to the second aeration tank, where the organic material adsorbed onto the biomass surface is metabolized or "stabilized" (Benefield and Randall, 1980).

Sludge Reaeration

Because of the aeration deficiencies resulted in an insufficient oxygen supply to the biomass, which therefore reduced the organic removal efficiency, many plants were designed to provide aeration of the return sludge. For the design of this modification, it is assumed that all substrate entering the reaeration tank is removed. Thus, no substrate will be present in the recycle from the reaeration tank to the aeration tank. With the aeration equipment available today, the original reason for using the sludge reaeration process is no longer valid (Benefield and Randall, 1980).

Pure Oxygen Aeration

The pure oxygen system produces less biomass and exhibits a higher substrate utilization rate than the air system. It is generally accepted that the pure oxygen has the following advantages over an air system; (1) capability to meet higher oxygen demands (2) ability to maintain a higher MLVSS concentration in the aeration tank and thus provide equivalent treatment in a smaller volume aeration tank (3) better sludge settling and thickening (4) lower net sludge production per unit BOD removed (5) can transfer more oxygen per horsepower (6) more stable treatment (Benefield and Randall, 1980).

Oxidation Ditches

The oxidation ditch is generally operated as an extended aeration process and employed for the treatment of small wastewater flows. It consists of an oval or "racetrack"-shaped channel about 1 m. deep. A brush or caged rotor aerator is placed across the ditch to aerate the mixed liquor as well as impart unidirectional flow to the passing liquid. The mixed liquor circulates in the ditch at a rate of 0.3-0.6 m/s. Of all the activated sludge process modifications, the oxidation ditch seems to be the least affected by operator skill (Benefield and Randall, 1980).

2.4 Activated Sludge Modifications for Biological Nutrient Removal

Later, when the need for nutrient treatment, along with the treatment of classical parameters such as COD and BOD, arose, activated sludge was found to be equally effective in doing this too. Several modifications have since been developed based on the design experience gained.

A/O (Anaerobic/Oxic) Process

The A/O process has two stages. That is, an anaerobic stage precedes an aerobic stage for enhanced biological phosphorus removal (BPR), as shown in Figure 2.2. The two-stage A/O process, developed primarily for carbon oxidation and phosphorus removal, is specifically designed to optimize enhanced biological phosphorus removal (EBPR). Typically, each of the anaerobic and aerobic stages are divided into a number of equally sized, completely mixed compartments. The clarifier underflow returns to the first compartment of the anaerobic zone (WEF and ASCE, 1992).

This process will likely to function most effectively when nitrification does not occur as the process lacks any provision for the removal of nitrates returned to the anaerobic zone with the return sludge. The process can be adapted for nitrification by allowing prolonged detention time in the aerobic stage but this may hamper the P-removal. A significant denitrification does not take place in this configuration as the

process lacks an anoxic zone for nitrate reduction. The key features of the A/O process are its relatively short SRT and high organic loading rates (Sedlak, 1991).

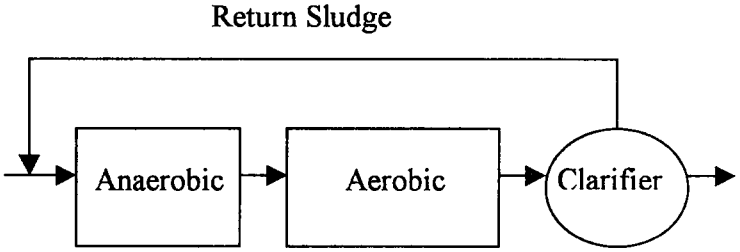

Figure 2.2 A/O Process

A^2/O (Anaerobic/Anoxic/Oxic) Process

The A^2/O process can also be used where nitrification and/or denitrification are required. The modified flow scheme incorporates an anaerobic stage followed by anoxic stage for denitrification and an aerobic stage as shown in Figure 2.3 (Sedlak, 1991).

In this configuration, mixed liquor is recycled from the aerobic nitrification stage into the anoxic stage to affect denitrification. Internal recirculation flows of 100-300 percent have been used. Clarifier underflow returns to the anaerobic first stage (WEF and ASCE, 1992). The nitrate-nitrogen removals of 40-70 percent can be achieved by this way (Sedlak, 1991).

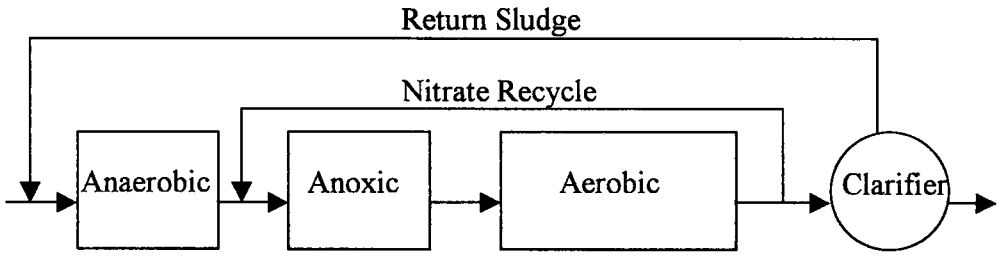

Figure 2.3 A^2/O Process

Bardenpho Process

The Bardenpho process for simultaneous carbon oxidation, nitrification, and denitrification can be depicted as four unequal sized tanks in series as shown in Figure 2.4. The tank no. 1 receives both recirculation of mixed liquor from tank 2 and the return sludge. In this flow scheme the first and third tanks are anoxic and the second and fourth tanks are aerobic to achieve nitrification and denitrification (Bidstrup and Grady, 1988).

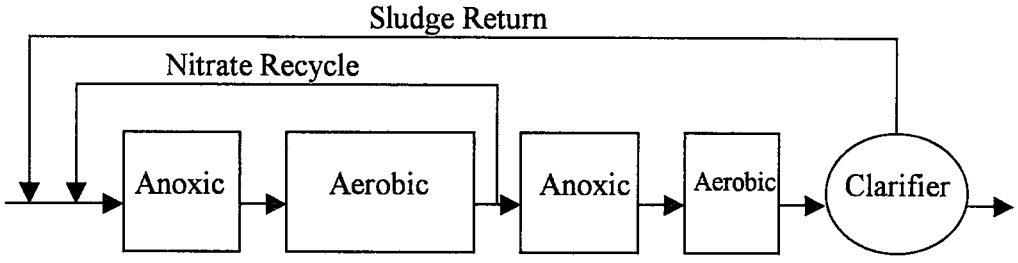

Figure 2.4 Bardenpho Process

2.5 Activated Sludge Kinetics

Simulation of the behaviour of activated sludge process involving multiple phenomena such as carbon oxidation, nitrification and denitrification, ought to involve a large number of components. To be mathematically tractable while providing realistic predictions, the reactions must be representative of the most important fundamental processes occurring within the system. Here the term *process* is used to mean a distinct event acting upon one or more system components. Furthermore, the model should quantify both the kinetics (rate-concentration dependence) and the stoichiometry (relationship that one component has to another in a reaction) of each process. Identification of the major processes and selection of the appropriate kinetic and stoichiometric expressions for each, are the major conceptual tasks during development of a mathematical model (Henze *et al.*, 1986).

2.5.1 Matrix Notation

It is often difficult to trace all the interactions of the system components as the model gets too complicated. The International Association on Water Pollution Research and Control (IAWPRC) Task Group (Henze *et al.*, 1986) concluded a matrix format for presentation of the model. The matrix approach offered the best opportunity for overcoming the complication problem while conveying the maximum amount of information. The following illustration should introduce to the matrix format and the relevant notation:

Let's consider the situation in which heterotrophic bacteria are growing in an aerobic environment by utilizing a soluble substrate for carbon and energy. In one simple conceptualization of this situation, two fundamental processes occur: the biomass increases by cell growth and decreases by decay. Other events, such as oxygen utilization and substrate removal, also occur, but these are not considered to be fundamental because they result from biomass growth and decay and are coupled to them through the system stoichiometry. The simplest model of this situation must consider the concentrations of three components: biomass, substrate, and dissolved oxygen. The matrix incorporating the fate of these three components in the two fundamental processes is shown in Table 2.1.

Table 2.1 Process kinetics and stoichiometry for heterotrophic bacterial growth in an aerobic environment (Henze *et al.*, 1986)

→ Continuity

Mass Balance ↓

Component W | i j Process X	1 X_B	2 S_S	3 S_O	Process Rate, ρ_j $[ML^{-3}T^{-1}]$
1 Growth	1	$-\frac{1}{Y}$	$-\frac{1-Y}{Y}$	$\frac{\hat{\mu}S_S}{K_S+S_S}X_B$
2 Decay	-1		-1	bX_B
Observed conversion rates $ML^{-3}T^{-1}$	$r_i = \sum_j r_{ij} = \sum_j \nu_{ij}\rho_j$			Kinetic parameters: Maximum specific growth rate: $\hat{\mu}$
Stoichiometric parameters: True growth yield factor : Y	Biomass $[M(COD)L^{-3}]$	Substrate $[M(COD)L^{-3}]$	Oxygen (negative COD) $[M(-COD)L^{-3}]$	Half-velocity constant: K_S Specific decay rate: b

Identification of the components of relevance in the model forms the first step in setting up the matrix. In this example these are biomass, substrate and dissolved oxygen, which are listed across the top of Table 2.1 by symbol and across the bottom by name and units. In conformity with IAWPRC nomenclature, insoluble constituents are given the symbol X and the soluble components S. Subscripts are used to specify individual components: B for biomass, S for substrate and O for oxygen. The index i is assigned to each component. In this case, i ranges from 1 to 3 for the three compounds in this simple model (Henze *et al.,* 1986).

Identification of the biological processes occurring in the system, i.e. the conversions or transformations, which affect the components, forms the second step in developing the matrix. In this example only two processes are included: aerobic growth of biomass and its loss by decay. These processes are listed in the leftmost column of the matrix. The index j is assigned to each process; in this case, j=1 or 2.

In the appropriate row of the rightmost column of the matrix, the kinetic expressions or rate equations for each process are given. Process rates are denoted by ρ_j, where j corresponds to the process as numbered in the leftmost column. If we were to use the simple Monod-Herbert model for this situation the rate expressions would be those given in Table 2.1. The Monod equation, ρ_1, says that growth of biomass is proportional to biomass concentration in a mixed order manner. The Herbert expression, ρ_2, states that biomass decay is first order with respect to biomass concentration. The kinetic parameters used in the rate expressions are defined in the lower right corner of the table (Henze *et al.,* 1986).

The elements within the matrix comprise the stoichiometric coefficients, v_{ij}, which set out the mass relationship between the components in the individual processes. For example, growth of biomass (+1) occurs at the expense of soluble substrate (-1/Y); oxygen is utilized in the metabolic process [-(1-Y)/Y]. The coefficients, v_{ij}, are greatly simplified by working in consistent units. In this case, all organic constituents have been expressed as equivalent amounts of chemical oxygen demand (COD); likewise, oxygen is expressed as negative oxygen demand. The sign convention used in the matrix is negative for consumption and positive for production. All stoichiometric coefficients are defined in the lower left corner of the table (Henze *et al.*, 1986).

Use in Mass Balances

The concentration of a single component may be affected by a number of different processes within a system. Matrix representation allows rapid and easy recognition of the fate of each component and this aids in the preparation of mass balance equations. The arrow marked 'Mass Balance' is placed at the left hand side of the column representing components. The basic equation for a mass balance within any defined system boundary is:

$$\text{Input} - \text{Output} + \text{Reaction} = \text{Accumulation} \qquad 2.4$$

The input and output terms are transport terms and depend upon the physical characteristics of the system being modeled. The system reaction term, r_i, is obtained

by summing the products of the stoichiometric coefficients v_{ij} and the process rate expression ρ_j, for the component i being considered in the mass balance:

$$r_i = \sum_j v_{ij} \rho_j \qquad 2.5$$

For example the rate of reaction, r, for biomass, X_B, at a point in the system would be:

$$r_{X_B} = \frac{\hat{\mu} S_S}{K_S + S_S} X_B - b X_B \qquad 2.6$$

For soluble substrate, S_S, it would be:

$$r_{S_S} = -\frac{1}{Y} \frac{\hat{\mu} S_S}{K_S + S_S} X_B \qquad 2.7$$

For dissolved oxygen, S_O, it would be:

$$r_{S_O} = -\left(\frac{1-Y}{Y}\right) \frac{\hat{\mu} S_S}{K_S + S_S} X_B - b X_B \qquad 2.8$$

In order to create the mass balance for each component within a given system boundary, the conversion rate would be combined with the appropriate advective terms for the particular system. Since the purpose of this example is to demonstrate how the matrix is used to define the fundamental reactions regardless of the system

configuration, these terms have not been included here. It should be emphasized, however, that modeling of a particular physical system requires definition of the system boundary with the associated advective terms (Henze *et al.*, 1986).

Continuity Check

Matrix formation presents another benefit as its continuity may be checked by moving across the matrix. Provided that consistent units have been used, the sum of stoichiometric coefficients must be zero. This can be demonstrated by considering the decay process. Since oxygen (S_O) is negative COD, its coefficient must be multiplied by –1. Biomass (X_B) decay coefficient is –1. All COD lost from the biomass must be equal to the oxygen used since decay must be balanced by oxygen utilization. Similarly, for the growth process, the substrate COD lost from solution due to growth minus the amount converted into new cells must equal the oxygen used for cell synthesis (Henze *et al.*, 1986).

2.5.2 Activated Sludge Model Number 1 (ASM1)

The International Association on Water Pollution Research and Control (IAWPRC) formed a Task Group on 'Mathematical Modeling for Design and Operation of Biological Wastewater Treatment' in 1983. This Task Group, consisting of one representative from each of five leading countries in activated sludge modeling, developed a consensus model that will predict the performance and realistically mimic the single activated sludge systems in which carbon oxidation, nitrification and denitrification are simultaneously achieved. The proposed model by the Task Group, named Activated Sludge Model No.1 (ASM1), is presented in Table 2.2.

It is known that the carbon and nitrogen removals are accomplished basically by two different groups of microorganisms. The first group is heterotrophic microorganisms. Heterotrophic microorganisms grow at the expense of soluble organic matter in aerobic and anoxic conditions. They can utilize nitrate as terminal electron acceptor under anoxic conditions. The other group is autotrophic microorganisms. Autotrophic microorganisms oxidize organic nitrogen to nitrate under aerobic conditions. These are the microorganisms carrying out nitrification (Henze *et al.*, 1986).

Within this context, it should be noted that the task group employed switching functions to turn process rate equations on and off as environmental conditions are changed. This was particularly necessary for the processes that depend upon the type of electron acceptor present.

Table 2.2 Process kinetics and stoichiometry for carbon oxidation, nitrification, and denitrification, ASM1 (Henze *et al.*, 1986)

Component	i	1	2	3	4	5	6	7	8	9	10	11	12	13	Process Rate ρ_I
j	Process	S_I	S_S	X_I	X_S	$X_{B,H}$	$X_{B,A}$	X_P	S_O	S_{NO}	S_{NH}	S_{ND}	X_{ND}	S_{ALK}	$(ML^{-3}T^{-1})$
1	Aerobic growth of heterotrophs		$-\frac{1}{Y_H}$			+1			$-\frac{1-Y_H}{Y_H}$		$-i_{XB}$			$-\frac{i_{XB}}{14}$	$\hat{\mu}_H\left(\frac{S_S}{K_S+S_S}\right)\left(\frac{S_O}{S_O+K_{O,H}}\right)X_{B,H}$
2	Anoxic growth of heterotrophs		$-\frac{1}{Y_H}$			+1				$-\frac{1-Y_H}{2.86\cdot Y_H}$	$-i_{XB}$			$\frac{1-Y_H}{14\cdot 2.86Y_H}$ $-\frac{i_{XB}}{14}$	$\hat{\mu}_H\left(\frac{S_S}{K_S+S_S}\right)\left(\frac{K_{O,H}}{S_O+K_{O,H}}\right)X_{B,H}$ $\cdot\left(\frac{S_{NO}}{K_{NO}+S_{NO}}\right)\eta_g X_{B,H}$
3	Aerobic growth of autotrophs						+1		$-\frac{4.57-Y_A}{Y_A}$	$+\frac{1}{Y_A}$	$-i_{XB}-\frac{1}{Y_A}$			$-\frac{i_{XB}}{14}-\frac{1}{7Y_A}$	$\hat{\mu}_A\left(\frac{S_{NH}}{K_{NH}+S_{NH}}\right)\left(\frac{S_O}{S_O+K_{O,A}}\right)X_{B,A}$
4	'Decay' of heterotrophs				$1-f_P$	-1		f_P					$i_{XB}-f_P i_{XP}$		$h_H X_{B,H}$
5	'Decay' of autotrophs				$1-f_P$		-1	f_P							$h_A X_{A,H}$
6	Ammonification of soluble organic nitrogen										+1	-1		$\frac{1}{14}$	$k_a S_{ND} X_{B,H}$
7	'Hydrolysis' of entrapped organics		1		-1										$k_h\frac{X_S/X_{B,H}}{K_X+X_S/X_{B,H}}\left(\left(\frac{S_O}{K_O+S_O}\right)+\eta_h\left(\frac{K_{O,H}}{K_{O,H}+S_O}\right)\cdot\left(\frac{S_{NO}}{K_{NO}+S_{NO}}\right)\right)X_{B,H}$
8	'Hydrolysis' of entrapped organic nitrogen											+1			$\rho_T(X_{ND}/X_S)$
Observed Conversion Rates, $(ML^{-3}T^{-1})$		$r_i=\sum_j r_{ij}=\sum_j \nu_{ij}\rho_j$													

For example the bacteria, which are responsible for nitrification are capable of growth only under aerobic conditions and their rate of growth will fall to zero as the dissolved oxygen concentration approaches zero, regardless of the concentration of their energy yielding substrate. This can be modeled by including dissolved oxygen 'switch' in the process rate equations. The oxygen switching function adopted by the Task Group was:

$$\frac{S_o}{K_o + S_o} \qquad 2.9$$

where S_O is the concentration of dissolved oxygen. Selecting a small value for K_O means that the value of the switching function is near unity for moderate dissolved oxygen (DO) concentrations and therefore would not cause a considerable influence on the rate expression. But the switching function decreases to zero as the DO concentration approaches zero; therefore growth rate of autotrophs, which are strict aerobes, will be zero in the absence of oxygen. Since the function is mathematically continuous, elimination of problems of numerical instability becomes possible. In a similar way, processes which occur when dissolved oxygen is absent may be turned on by a switching function of the form:

$$\frac{K_o}{K_o + S_o} \qquad 2.10$$

It is obvious that the values of switching constants like K_O have significant influence on rate expressions and thus predictions of the activated sludge concentrations are highly affected by these constants. Task Group adopted switching functions for their mathematical convenience rather than their conformity with the fundamental rate laws. This is an important point that should be considered during selection of these values (Henze *et al.*, 1986).

Conceptual Model

The Task Group used the chemical oxygen demand (COD) for the measure of the organic matter in wastewater since it not only provides a link between equivalents of electrons in organic substrate, the biomass and the oxygen utilized; but it also provides a basis for mass balance within the system.

In the Activated Sludge Model No.1, organic matter in the wastewater is separated into two groups depending on their biodegradability. First group is non-biodegradable organic matter, which is biologically inert and leaves the system at the same form as it enters. This group is separated into two groups depending on their physical state as inert soluble organic matter, S_I which passes through the system at the same concentration and inert suspended organic matter, X_I , concentration of which can increase since it is trapped by the activated sludge flocs. Second group is biodegradable organic matter, which can be divided into two groups as readily biodegradable substrate, S_S, which is directly used by heterotrophic bacteria for growth and treated. The slowly biodegradable material, X_S, which is converted to

readily biodegradable substrate after hydrolysis and treated as if particulate (Henze *et al.*, 1986).

The specific utilization rate is generally considerably higher than the specific hydrolysis rate of readily biodegradable organic matter. Because of this, hydrolysis becomes a rate-limiting process during growth of biomass on X_S. Hydrolysis rate decreases under anoxic conditions, as compared to aerobic conditions. The only terminal electron acceptor is nitrate in anoxic conditions and hydrolysis completely stops under anaerobic conditions (Henze *et al.*, 1986).

It is assumed that heterotrophic biomass growth takes place only under aerobic and anoxic conditions at the expense of readily biodegradable substrate and completely stops under anaerobic conditions. The loss of biomass within the system is modeled by decay concept. Decay concept includes endogenous metabolism, predation, death and lysis. Decay results in the conversion of biomass into slowly biodegradable substrate and particulate products, X_P which cannot be further degraded (Henze *et al.*, 1986).

Like carbonaceous matter, nitrogenous matter in the wastewater can be separated into two groups depending on their degradability. Non-biodegradable fraction is the particulate fraction and associated with the non-biodegradable organic matter. The biodegradable fraction is usually small and not included in the model. Biodegradable fraction includes ammonia, S_{NH}, soluble organic nitrogen, S_{ND}, and particulate organic nitrogen X_{ND} which is hydrolyzed to soluble organic nitrogen. The ammonia

nitrogen serves as the nitrogen source for heterotrophic biomass and serves as the energy source for autotrophic nitrifying bacteria for growth. Autotrophic biomass concentration in the wastewater is not considered by the Task Group (Henze *et al.*, 1986).

Components in the ASM1

Moving through the columns in Table 2.2, components of ASM1 can be explained briefly. Columns i=1 and 3, contain no stoichiometric coefficients since soluble inert and particulate organic matter, S_I and X_I, are not involved in any conversion processes. In column i=2, it can be seen that readily biodegradable substrate, S_s, is removed by growth of heterotrophic bacteria under either aerobic or anoxic conditions and is formed by hydrolysis of particulate organic matter entrapped in the biofloc. The i=4 column reveals that slowly biodegradable substrate X_S, is removed by hydrolysis but is formed by decay of both heterotrophic and autotrophic biomass. The column i=5 and 6 represent the biomass in the system, with $X_{B,H}$ denoting the heterotrophic biomass and $X_{B,A}$ the autotrophic biomass. The i=7 column contains the particulate products arising from biomass decay, X_P.

The i=8 column contains the concentration of DO, S_O, in the reactor. The processes included in the matrix act to remove oxygen from solution but they do not act to add. For the simulation of DO concentration, process rate expressions for oxygen transfer would have to be included with the transport terms when writing the mass balance equation for oxygen. The factor of 4.57 in the stoichiometric coefficient for aerobic

growth of autotrophs is the theoretical oxygen demand associated with the oxidation of ammonia nitrogen to nitrate nitrogen. The other electron acceptor included in the model is nitrate nitrogen, S_{NO}, as can be seen in the i=9 column. The factor 2.86 in the stoichiometric coefficient for anoxic growth of the heterotrophic biomass is the oxygen equivalence for conversion of nitrate nitrogen to nitrogen gas (N_2).

The i=10 column contains soluble ammonia nitrogen, S_{NH}, which is assumed to be the sum of the ionized (ammonium) and un-ionized (ammonia) forms. The i=11 column contains the soluble organic nitrogen, S_{ND}, which is formed by hydrolysis of particulate organic nitrogen and converted to ammonia nitrogen by ammonification. Particulate biodegradable organic nitrogen, X_{ND}, is given in the i=12 column. Three other forms of organic nitrogen will be present in the system: that associated with the biomass, X_{NB}; that associated with the particulate products, X_{NP}; and that associated with the inert particulate organic matter, X_{NI}. These 12 components discussed are considered to be the minimum required ones to model adequately an activated sludge system performing carbon oxidation, nitrification, and denitrification. The i=13 column represents total alkalinity, S_{ALK}. Incorporation of alkalinity into the model is not essential.

Processes in the ASM1

Basically, four processes are considered in the model: growth of biomass, decay of biomass, ammonification of organic nitrogen, and hydrolysis of particulate organics.

Growth of biomass is considered separately for autotrophic and heterotrophic microorganisms under aerobic and anoxic conditions. Aerobic growth of heterotrophic biomass occurs at the expense of readily biodegradable substrate and results in the production of heterotrophic biomass. Utilization of oxygen is associated with this. In anoxic growth of the heterotrophic biomass nitrate nitrogen is used as the terminal electron acceptor. Like aerobic growth it occurs at the expense of readily biodegradable substrate and results in heterotrophic biomass. In aerobic growth of autotrophic biomass, soluble ammonia nitrogen serves as the energy source for growth of the nitrifiers resulting in autotrophic cell mass and nitrate nitrogen as end products. In addition, a small amount of ammonia nitrogen is oxidized.

Decay of biomass is considered separately for autotrophic and heterotrophic microorganisms also. Decay is assumed to result in the transformation of active biomass into inert particulate products and into slowly biodegradable substrate, which re-enters the cycle of hydrolysis, growth, etc. This allows straightforward expression of decay under the various environmental conditions encountered in a single sludge system. It is well known that the observed yield from the growth of heterotrophic biomass decreases as the SRT of a reactor is increased. This phenomenon is thought to be due to many mechanisms, including predation, lysis, and the need for maintenance energy. General approach during their modeling is to collect them with one expression called decay. The approach adopted for modeling decay of the heterotrophic biomass is basically the death-regeneration concept. The adopted rate expression is first order with respect to the heterotrophic biomass concentration. In this case, decay acts to convert biomass to a combination of

particulate products and slowly biodegradable substrate. No loss of COD is involved in this split and no electron acceptor is utilized. Furthermore, decay continues at a constant rate regardless of the environmental conditions. The decay of autotrophs, is handled in exactly the same manner as the decay of heterotrophs.

Soluble organic nitrogen is converted to ammonia nitrogen through the reaction depicted in row 6 of Table 2.2. Equation presented here for the ammonification of soluble organic nitrogen is a first order equation, which is found to be adequate for modeling the conversion.

Hydrolysis process is modeled for entrapped organics and entrapped organic nitrogen. Slowly biodegradable material is considered to be removed from suspension by entrapment in the biofloc. Once it reaches into the biofloc, reactions start which will convert it into readily biodegradable substrate. These reactions are simply called hydrolysis in the model, although in reality they are likely to be much more complex. In the model, the organic nitrogen is thought to be uniformly distributed throughout the slowly biodegradable substrate. Therefore, the rate of hydrolysis of entrapped organic nitrogen would simply be proportional to the rate of hydrolysis of slowly biodegradable substrate (Henze *et al.*, 1986).

2.6 Population Dynamics in Activated Sludge

2.6.1 Historical Development

The objective of modeling activated sludge population dynamics is to bring together information governing biological wastewater treatment. Activated sludge models are also useful in testing theories on the biological processes involved as well as developing new theoretical description of such systems. New wastewater characterization procedures may be sought with the help of models, as these may apply to different array of microorganisms making up the biological flocs, hence promoting understanding and modeling of solids separation. An additional goal is to contribute to the building of an activated sludge model that can predict "activated sludge quality" and the behavior of activated sludge in gravity clarifiers (Jenkins, 1992).

Solids Separation Problems

It is well known that the performance of a gravity secondary clarifier has a major influence on the final effluent quality. It also has a great importance on the sizing of aeration basin and solids processing units and their performance through its ability to produce a concentrated underflow (Jenkins, 1992).

Solids separation problems have been identified by some common names and with their causes. Contrary to general belief, not all solids separation problems are caused

by the filamentous organisms. Indeed, only filamentous bulking and one type of foaming or floating sludge may be directly related to the presence of excessive amounts of filamentous organisms. Conversely pin-point floc formation and effluent turbidity problems may be related to low filamentous organism levels. Hence, this research is confined to modeling of filamentous-organisms' behavior in activated sludge.

Most activated sludge cultures contain some degree of filamentous organisms. Their presence within activated sludge flocs at low levels do not create problems; actually low levels of filamentous organisms provide structure and strength to activated sludge flocs (Parker *et al.*, 1971). Another belief is that filamentous organisms are not needed to provide strong flocs (Chudoba, 1989). A properly formed and tightly bound glycolyx is presumed to confer floc strength. The two theories on floc strength are not mutually exclusive. In fact, viscous bulking can occur in the total absence of filamentous microorganisms. Whereas in most of the cases, and certainly in domestic effluent treatment, floc strength is conferred by the presence of filamentous organisms.

The amount of filamentous organism density required for proper flocs is reported as 10^7 μm/g VSS (Sezgin *et al.*, 1978). Lesser filament concentrations is reported to cause deterioration in activated sludge settling properties and turbid effluent. Whereas filament densities significantly higher than 10^7 μm/g VSS occur at low DO concentrations and the typical filamentous microorganisms are *Sphaerotilus natans* or type 1701 (Sezgin *et al.*, 1978). Since various filamentous organisms have a range

of growth forms where rigid straight trichomes stick out into the bulk solution but coiled filaments wrap around the flocs, it is to be expected that the effect on settling properties will also change by the type of filamentous organism. It is estimated that a bulking sludge would occur when the extended filamentous organisms volume exceeded approximately 1% of the total floc volume (Palm *et al.*, 1980).

Estimates like these cannot be made for *Nocardia* spp or *Microthrix parvicella* which are known to cause nuisance foam in activated sludge plants. It is because reactor configuration and physical/chemical properties of solution are also important in determining foam levels. For example presence of poorly biodegradable surfactants may cause foaming or a poorly designed aeration tank may lead to trapping of foam in this basin.

The culture composition of sludge would also be responsible of producing nuisance foams. Nuisance foams have been found when the *Nocardia* counts in the mixed liquor is in the range greater than 10^6 intersections/g VSS which corresponds to approximately $\geq$ 26 mg Nocardia/g VSS, or $\geq$3% by weight of the activated sludge culture (Ho and Hernandez, 1991).

Both scenarios are indicative of settling, bulking or foaming problems which occur when responsible filamentous organisms make up very small percentages of the activated sludge floc population. This suggested that a model predicting conditions that favor or disfavor filamentous organisms and hence their relative counts within flocs should be useful in forecasting bulking or foaming situations (Jenkins, 1992).

Groups of Filamentous Organisms

Several groups of filamentous organisms, which relate to different bulking situations, may be distinguished (Jenkins, 1992).

1. Filaments such as *S. natans*, Type 021N, Type 0961. They are typically found in aerobic systems, which have a solids retention time less than 10 days. These filaments extend out of the flocs into the wastewater. This group of filaments is dominant in most Swiss wastewater treatment plants (Gujer and Kappeler, 1992).

2. Filaments such as *M. parvicella* and Type 0092. They are typically found in systems with high solids retention time. These filaments grow within the flocs and may use the substrates generated inside the flocs by hydrolysis (Gujer and Kappeler, 1992).

3. Filaments such as *Beggiaota* and *Thiothrix*. These filaments exist in the presence of sulfides (Gujer and Kappeler, 1992).

4. Actinomycetes. These filaments adsorb on to gas bubbles and float on the surface of either the aeration tank or secondary clarifier; where they may form a scum layer, if scum is retained in the system (Gujer and Kappeler, 1992).

2.6.2 Modeling Population Dynamics

The present day complicated mathematical models which can simulate the complex activated sludge process, together with supporting software, are applied worldwide in the design and optimization of biological wastewater treatment systems. These models usually describe only the underlying processes and do not predict the relative

abundance of groups of organisms. Latter is termed "population dynamics" (Gujer and Kappeler, 1992).

Models, which can predict settling characteristics of activated sludge are still at an early phase of development. Some qualitative models, which can predict the presence and absence of filamentous organisms following selectors, are available. These models are usually too simple to come close enough to reality in order to be applied in design work (Gujer and Kappeler, 1992).

Simple Models from the Literature

Several qualitative models are reported in the literature. These models have different disadvantages and would not usually be applicable for the simulation of domestic wastewater treatment. They may however give a qualitative insight into the dynamics of the biomass composition in activated sludge systems (Gujer and Kappeler, 1992).

Two Organisms Growing on Different Substrates

If two organisms are considered, each of which feeding on a different substrate, no competition exists for common substrates and it is primarily the relative abundance of the two substrates in the influent which defines the composition of the activated sludge biocenoses. A selector or any other reactor in the flow scheme could not affect this biocenoses significantly (Gujer and Kappeler, 1992).

Growth of autotrophic nitrifying organisms within a heterotrophic biomass can be given as an example to this case. As long as adequate oxygen is supplied for the development of both groups of organisms, the design of the reactor barely affects activated sludge composition (Gujer and Kappeler, 1992).

It is obvious that this model is not useful for the prediction of the relative abundance of filamentous and floc-forming heterotrophic organisms (Gujer and Kappeler, 1992).

Two Organisms Growing on a Single Soluble Substrate

Two types of organisms are described: (i) those who rely on an r-strategy, which compete on their high rates of activity, and characterized by a high maximum specific growth rate, μ_m and (ii) those who rely on K-strategy, which compete on their high substrate affinity or low requirements for substrate concentration, as characterized by a low K_S value in Monod model (Gujer and Kappeler, 1992).

If these two types of organisms, floc-formers and filamentous, compete for the same soluble substrate at steady state, usually one of the two organisms is washed out of the system. This case results in a biocenoses, which is either all filamentous or all floc-formers, depending on the mixing regime of the aeration tank. In a selector with high F/M ratio, a growth environment providing high substrate concentration is created. This environment will enrich organisms which rely on r-strategy. Whereas in a completely mixed reactor, organisms relying on a K-strategy would have

advantage. General assumption is that floc-formers rely on r-strategy whereas filaments on K-strategy (Gujer and Kappeler, 1992).

This model has the disadvantage of considering only one type of substrate, which is soluble. The model overestimates oxygen consumption in the selector compartments because of the immediate degradation of the soluble substrate. It only takes into account one species of organisms. Both observations cannot be experimentally verified in domestic wastewater treatment (Gujer and Kappeler, 1992).

Floc-forming Organisms with Substrate Accumulation Capacity

Floc-forming organisms can store considerable amounts of substrate whereas filamentous organisms cannot. The removal of substrate as long as the accumulation capacity is not saturated, is quite fast (Gujer and Kappeler, 1992). In an aerobic selector, floc formers having a high soluble substrate carbon storage capacity and higher soluble carbon uptake rate will dominate over filamentous microorganisms under unbalanced growth conditions (Jenkins, 1992).

This model was supposed to explain a great variety of different phenomena occurring in context of bulking activated sludge. This is a more complex version of the previously mentioned model and additionally includes two types of substrates for heterotrophic growth: (i) the soluble material fed to the system with the influent and (ii) the substrate accumulated within the flocforming organisms. It presents an advantage as it makes possible to predict the consumption of oxygen along a plug

flow type reactor. However, it has a disadvantage since it will usually predict a single culture of microorganisms for any flow scheme and operating conditions (Gujer and Kappeler, 1992).

It is known that current qualitative models designed for the prediction of filamentous and floc-forming organisms in activated sludge are based on a single type of organic substrate and a single type of organism in wastewaters and generally the predicted oxygen consumption at the head end of a plug flow type aeration tank is always unrealistically high. Hence, a more representative model should rely on the use of several groups of organic substrates, which would allow prediction of activated sludge composition with more than one type of organisms (Gujer and Kappeler, 1992).

2.7 AEROFIL Model for Modeling Population Dynamics

A mathematical model describing behavior of facultative aerobic floc-forming, as well as obligate aerobic filamentous and nitrifying microorganisms, was developed. It was hypothesized that competition between floc-forming and filamentous organisms requires distinction of readily biodegradable substrates in the influent from readily biodegradable hydrolysis products formed during the process. The distinction was based on the different diffusional resistances that each group of organic matter possesses. Kinetic parameters of the heterotrophic floc-forming and the nitrifying microorganisms can directly be estimated by batch-tests. Obligate aerobic filamentous microorganisms only represent a small fraction of the biocenoses

therefore their kinetic parameters must be estimated with indirect methods (Kappeler and Gujer, 1994a).

Bulking and scum accumulation are common phenomena in activated sludge plants. A considerable percentage of the wastewater treatment plants from time to time suffer from bulking or scumming due to excessive growth of filamentous microorganisms. To discuss the problems related to filamentous microorganisms, it is helpful to distinguish the following four main functional groups:

- Aerobic bulking (*Sphaerotilus natans*, Type 021N, Type 0961,...)
- Scumming due to Actinomycetes (e.g. *Nocardia* sp.)
- Low F/M bulking and scumming (*Microthrix parvicella*, Type 0092, Type 0041,...)
- Bulking due to sulphide oxidizing bacteria (Thiothrix, Beggiatoa, Type 021N).

Scumming due to Actinomycetes may be avoided by using anoxic selectors. There is no well defined control methods known for low F/M bulking and scumming. Bulking due to sulphide oxidizing bacteria can be avoided by eliminating the sulphide source (Kappeler and Gujer, 1994a).

In the case of aerobic bulking, excessive growth of filamentous microorganisms can be avoided by using aerobic selectors. The mathematical simulation model AEROFIL (Kappeler and Gujer, 1994a) was developed for this problem.

Filamentous Microorganisms Causing Aerobic Bulking

The filamentous microorganisms responsible for aerobic bulking are commonly *Sphaerotilus natans*, Type 021N and Type 0961. S*phaerotilus natans* is obligate aerobic, though it has a very low oxygen half-saturation coefficient. *Sphaerotilus natans* and Type 021N did not grow under anoxic conditions (Wanner *et al.*, 1987). Type 0961 was washed out of the activated sludge of a sequencing batch reactor operated in an anoxic and then aerobic cycle. Type 0961 was not washed out from a completely aerobic sequencing batch reactor (Kappeler and Gujer, 1994a). Based on these studies it can be concluded that the filamentous microorganisms which are responsible for aerobic bulking are obligate aerobic, whereas most of the floc-forming microorganisms are considered to be facultative aerobic.

2.7.1 Development of the AEROFIL Model

AEROFIL Model aims to describe the behavior of a biocenoses consisting of facultative aerobic floc-forming organisms along with the obligate AERObic FILamentous and nitrifying microorganisms under various circumstances. AEROFIL model is restricted to the following validity range, if the biocenoses composition is to be predicted:

- Aerobic systems, having a solids retention time less than 10 days.
- Aerobic systems with a small anoxic zone, solids retention time less than 10 days.

- Aerobic systems with a small anaerobic zone which does not employ biological phosphorus removal; there is no nitrification because of the reduced solids retention time.
- No amounts of surfactants which may favor the proliferation of Actinomycetes, or long chain fatty acids which may favor the proliferation of *Microthrix parvicella*,

When these model limitations are exceeded, the AEROFIL model may not be useful. In such a case the most probable biocenoses composition cannot be predicted any more, since other groups of filamentous microorganisms such as Actinomycetes or *Microthrix parvicella* may gain importance (Kappeler and Gujer, 1994a).

The activated sludge model No.1 (Henze *et al.*, 1986) is based on two different types of organic substrates as it was mentioned before. Influent COD is assumed to be fractioned into two groups: soluble and readily biodegradable and colloidal to particulate slowly biodegradable. Slowly biodegradable fraction is assumed to undergo cell external hydrolysis before it may be degraded. Activated sludge model No.1 does not include any distinction between influent readily biodegradable COD and the products of the hydrolysis of slowly biodegradable COD. In the AEROFIL Model it is suggested that there might well be a very significant physical rather than chemical difference between the two groups of substrates.

Readily Biodegradable COD from the Influent

The readily biodegradable influent COD, S_S, is introduced to the reactor via the influent and is available in the outside of the activated sludge flocs, possibly in direct contact with many filaments reaching out of the flocs into the surrounding water. Readily biodegradable COD must first diffuse into the flocs in order to be available to floc-forming organisms. A diffusional resistance may be modeled with Monod kinetics, by increasing the apparent saturation constant K_S above its intrinsic value K_{SO}. Therefore it is expected that filamentous organisms have a low and floc-forming organisms a high apparent saturation constant (K_S) for influent readily biodegradable COD (Kappeler and Gujer, 1994a).

Readily Biodegradable COD Released by Hydrolysis

Particulate and colloidal organic materials are hydrolyzed into readily biodegradable COD. Hydrolysis is a cell external phenomenon and it is not known what fraction of hydrolysis products S_H ever exists in true solution. Particulate material in activated sludge is associated with the flocs and is adsorbed onto filamentous organisms, which are responsible for aerobic bulking. This indicates that hydrolysis takes place within the activated sludge flocs and hydrolysis products become primarily available to floc-forming organisms. Since there is only little diffusion resistance between the site of production and the site of consumption of these substrates, the value of the apparent saturation constant K_H for floc-forming organisms is expected to be low. Since hydrolysis products must diffuse out of the flocs in order to become available

to filamentous organisms, the apparent K_H value for these organisms must be high (Kappeler and Gujer, 1994a).

Other COD fractions characterizing the wastewater according to the activated sludge model-ASM No. 1 are soluble inert COD S_I, slowly biodegradable organic matter X_S and inert suspended COD X_I. Three groups of microorganisms were included in the biokinetic model: facultative aerobic floc-formers X_{Floc}, obligate aerobic filaments X_{Fil} and nitrifiers X_{Nitr} in order to describe the competition between facultative aerobic floc-forming, and obligate aerobic filamentous and nitrifying microorganisms

Nitrification and denitrification are also described by the AEROFIL model. Hence, ammonium S_{NH4} and nitrate S_{NO3} were also included in the model where nitrogen products other than ammonium and nitrate were not considered, since these were not within the aim of the AEROFIL model. They are only needed as information in the case of dissolved oxygen, S_{O2}, absence whether nitrate is available as the electron acceptor for heterotrophic floc-formers (Kappeler and Gujer, 1994a).

AEROFIL model stoichiometry is presented in Table 2.3 and the process kinetics is presented in Table 2.4. The AEROFIL model includes aerobic and anoxic growth of heterotrophic floc-forming microorganisms on readily biodegradable COD of the influent S_S, and on the soluble hydrolyzed products S_H. Since the filamentous microorganisms are assumed to be obligate aerobes, they can only grow under aerobic conditions. Nitrification is considered as a single step process from

ammonium to nitrate as it is in the activated sludge model No. 1. All decay processes of the different groups of bacteria are modeled as endogenous respiration under aerobic conditions and as lysis when oxygen is absent (Kappeler and Gujer, 1994a).

In the AEROFIL model, formulation of the hydrolysis process is different than it is in the activated sludge model No. 1 (Henze *et al.*, 1986). Hydrolysis term of the activated sludge model No. 1 is more complex but it did not improve the quality of the interpretation of the measured data. Therefore a more simple first order reaction relative to particulate biodegradable substrate X_S was chosen for the hydrolysis process. It is assumed that hydrolysis only takes place under aerobic conditions. It occurs under anoxic conditions with a reduced process rate, but not in anaerobic zones (Kappeler and Gujer, 1994a).

Modification in the AEROFIL Model Carried out in this Study

The AEROFIL model was slightly modified in this study to include aeration conditions within the plant configuration, as follows: For aerobic and anoxic conditions the following expressions were added to the model:

For aerobic conditions: $(10000*(SO_{2aer}-SO_2))$,

For anoxic conditions: $(KX_{O2}*(SO_{2sat}-SO_2))$

Here, SO_2 is the actual concentration of dissolved oxygen in the reactors. SO_{2aer} is the oxygen concentration that is achieved by a controlled reaeration process in aerobic reactors. It is assumed that the reaeration intensity automatically adjusts in order to achieve this concentration in a way that it is increased if the concentration

falls below this value and decreased if it is above. SO_{2sat} is the saturation concentration at a given temperature. This is the concentration, which the uncontrolled aeration process in anoxic reactor would achieve if oxygen would not be strongly consumed. "10000" is a very high rate constant ensuring the dissolved oxygen concentrations to stay at SO_{2aer} in aerobic reactors. The expression was added to keep a 2 g O/m^3 oxygen concentration in the aerobic tanks. It has no intrinsic physical meaning. It is chosen so large that the concentration does not significantly deviate from SO_{2aer}. K_{XO2} is aeration coefficient (1/d) (homologous to K_{LA}). The KX_{O2} value is normally taken as very low (5 1/d) to represent anoxic conditions. This is surface reaeration without any reaeration facility, which leads to an anoxic reactor.

For the excess sludge removal, the stoichiometric conversion factor was added as 1/SRT in the model. This converts growth expression to sludge wastage rate per unit volume of reactor.

Table2.3 Stoichiometry of the AEROFIL model (Kappeler and Gujer, 1994a)

Processes	S_{O_2}	S_S	S_H	S_I	X_S	X_I	X_{Floc}	X_{Nitr}	X_{Fil}	S_{NH_4}	S_{NO_3}
Facultative aerobic floc-formers X_{Floc}											
Aerobic growth on S_S	$-\frac{1-Y_H}{Y_H}$	$-\frac{1}{Y_H}$					+1			$-i_{XB}$	
Aerobic growth on S_H	$-\frac{1-Y_H}{Y_H}$		$-\frac{1}{Y_H}$				+1			$-i_{XB}$	
Anoxic growth on S_S		$-\frac{1}{Y_H}$					+1			$-i_{XB}$	$-\frac{1-Y_H}{2.86Y_H}$
Anoxic growth on S_H			$-\frac{1}{Y_H}$				+1			$-i_{XB}$	$-\frac{1-Y_H}{2.86Y_H}$
End. respiration	$-(1-f_P)$					$+f_P$	-1			$i_{XB}(1-f_P)$	
Lysis			$1-f_I$			$+f_I$	-1			$i_{XB}(1-f_I)$	
Nitrifiers X_{Nitr}											
Growth	$-\frac{4.57-Y_A}{Y_A}$							+1		$-i_{XB}-\frac{1}{Y_A}$	$\frac{1}{Y_A}$
End. respiration	$-(1-f_P)$					$+f_P$		-1		$i_{XB}(1-f_P)$	
Lysis			$1-f_I$			$+f_I$		-1		$i_{XB}(1-f_I)$	

Table 2.3 (continued) Stoichiometry of the AEROFIL model (Kappeler and Gujer, 1994a).

Processes	S_{O_2}	S_S	S_H	S_I	X_S	X_I	X_{Floc}	X_{Nitr}	X_{Fil}	S_{NH_4}	S_{NO_3}
Obligate aerobic filaments X_{Fil}											
Aerobic growth on S_S	$-\frac{1-Y_H}{Y_H}$	$-\frac{1}{Y_H}$							+1	$-i_{XB}$	
Aerobic growth on S_H	$-\frac{1-Y_H}{Y_H}$		$-\frac{1}{Y_H}$						+1	$-i_{XB}$	
End. respiration	$-(1-f_P)$					$+f_P$			-1	$i_{XB}(1-f_P)$	
Lysis		$1-f_I$				$+f_I$			-1	$i_{XB}(1-f_I)$	
Hydrolysis											
Aerobic			$1-f_H$	f_H	-1					i_{XS}	
Anoxic			$1-f_H$	f_H	-1					i_{XS}	
Aeration											
Aerobic	1										
Anoxic	1										
Sludge removal											
Excess sludge removal					$-X_S$	$-X_I$	$-X_{Floc}$	$-X_{Nitr}$	$-X_{Fil}$		
Units	COD	COD	COD	COD	COD	COD	COD	COD	COD	N	N

Table 2.4 Process kinetics of the AEROFIL model (Kappeler and Gujer, 1994a)

Process	Process rate
Facultative aerobic floc-formers X_{Floc}	
Aerobic growth on S_S	$\mu_{max,Floc} \cdot \left(\frac{S_S}{K_S + S_S} \cdot \frac{S_S}{S_S + S_H}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot X_{Floc}$
Aerobic growth on S_H	$\mu_{max,Floc} \cdot \left(\frac{S_H}{K_H + S_H} \cdot \frac{S_H}{S_S + S_H}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot X_{Floc}$
Anoxic growth on S_S	$\eta_g \cdot \mu_{max,Floc} \cdot \left(\frac{S_S}{K_S + S_S} \cdot \frac{S_S}{S_S + S_H}\right) \cdot \left(\frac{S_{NO_3}}{K_{NO_3} + S_{NO_3}}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot \left(\frac{K_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot X_{Floc}$
Anoxic growth on S_H	$\eta_g \cdot \mu_{max,Floc} \cdot \left(\frac{S_H}{K_H + S_H} \cdot \frac{S_H}{S_S + S_H}\right) \cdot \left(\frac{S_{NO_3}}{K_{NO_3} + S_{NO_3}}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot \left(\frac{K_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot X_{Floc}$
Endogenous respiration	$\eta_{endog} \cdot b'_{Floc} \cdot X_{Floc} \cdot \left(\frac{S_{O_2}}{K_O + S_{O_2}}\right)$
Lysis	$b'_{Floc} \cdot X_{Floc} \cdot \left(\frac{K_{O_2}}{K_{O_2} + S_{O_2}}\right)$
Nitrifiers X_{Nitr}	
Growth	$\mu_{max,Nitr} \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot X_{Nitr}$
Endogenous respiration	$\eta_{endog} \cdot b'_{Nitr} \cdot X_{Nitr} \cdot \left(\frac{S_{O_2}}{K_O + S_{O_2}}\right)$
Lysis	$b'_{Nitr} \cdot X_{Nitr} \cdot \left(\frac{K_{O_2}}{K_{O_2} + S_{O_2}}\right)$
Obligate aerobic filaments X_{Fil}	
Aerobic growth on S_S	$\mu_{max,Fil} \cdot \left(\frac{S_S}{K_S + S_S} \cdot \frac{S_S}{S_S + S_H}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot X_{Fil}$
Aerobic growth on S_H	$\mu_{max,Fil} \cdot \left(\frac{S_H}{K_H + S_H} \cdot \frac{S_H}{S_S + S_H}\right) \cdot \left(\frac{S_{NH_4}}{K_{NH_4} + S_{NH_4}}\right) \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot X_{Fil}$
Endogenous respiration	$\eta_{endog} \cdot b'_{Fil} \cdot X_{Fil} \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right)$
Lysis	$b'_{Fil} \cdot X_{Fil} \cdot \left(\frac{K_{O_2}}{K_{O_2} + S_{O_2}}\right)$
Hydrolysis	
Aerobic	$k_h \cdot X_S \cdot \left(\frac{S_{O_2}}{K_{O_2} + S_{O_2}}\right)$
Anoxic	$\eta_h \cdot k_h \cdot X_S \cdot \left(\frac{K_{O_2}}{K_{O_2} + S_{O_2}}\right) \cdot \left(\frac{S_{NO_3}}{K_{NO_3} + S_{NO_3}}\right)$
Units	$g \cdot m^{-3} \cdot d^{-1}$

2.7.2 Estimation of Kinetic Parameters

Kinetic parameters for the biokinetic AEROFIL model are listed in Table 2.5. These kinetic parameters represent typical values for treatment plants treating wastewater of domestic origin at 20°C. The basis for parameter identification is given below (Kappeler and Gujer, 1994a).

Table 2.5 Kinetic parameters (20°C) (Kappeler and Gujer, 1994a)

Kinetic parameters	*Symbol*	X_{Floc}	X_{Nitr}	X_{Fil}	*Units*
Maximum specific growth rate	μ_{max}	3.0	0.9	3.0	d^{-1}
Half-saturation coefficient for S_S	K_S	8	-	1	$gCOD/m^3$
Half-saturation coefficient for S_H	K_H	1	-	4	$gCOD/m^3$
Half-saturation coefficient for O_2	K_{O2}	0.25	0.4	0.1	gO_2/m^3
Half-saturation coefficient for NO_3	K_{NO3}	0.5	-	-	$gNO_3\text{-}N/m^3$
Coefficient for reduced anoxic growth	η_g	0.8	-	-	-
Half-saturation coefficient for NH_4	K_{NH4}	0.01	0.4	0.01	$gNH_4\text{-}N/m^3$
Yield coefficient of X_{Floc} and X_{Fil}	Y_H	0.63	-	0.63	gCOD/gCOD
Yield coefficient of X_{Nitr}	Y_A	-	0.24	-	gCOD/gN
Coefficient for lysis	b'	0.5	0.14	0.5	d^{-1}
Fractional coefficient for the release of inert particulate COD by lysis	f_I	0.08	0.08	0.08	gCOD/gCOD
Correction factor for endogenous respiration	η_{endog}	0.4	0.4	0.4	-
Fractional coefficient for the release of inert particulate COD by endogenous respiration	f_P	0.20	0.20	0.20	gCOD/gCOD
Ratio of nitrogen to COD in biomass and inert particulate COD	i_{XB}	0.06	0.06	0.06	gN/gCOD
Hydrolysis constant under aerobic conditions	k_h	5			d^{-1}
Fractional coefficient for the release of soluble inert COD by hydrolysis	f_H	0.07			gCOD/gCOD
Coefficient for reduced anoxic hydrolysis	η_h	0.6			-
Fraction of released ammonium by hydrolysis	i_{XS}	0.04			gN/gCOD

Kinetic parameters like μ_{max}, b′ and k_h are significantly influenced by the temperature. Temperature dependencies are shown in Table 2.6. The chosen dependencies are taken from Kappeler and Gujer, 1994b

Table 2.6 Temperature dependency of μ_{max}, b′ and k_h (Other kinetic parameters not significantly influenced by temperature) (Kappeler and Gujer, 1994b)

$c(T) = c(20^\circ C) \cdot e^{[\theta \cdot (T-20^\circ C)]}$					
		X_{Floc}	X_{Nitr}	X_{Fil}	Dimensions
θ for μ_{max}	Maximum specific growth rate	0.069	0.110	0.069	$(^\circ C)^{-1}$
θ for b′	Lysis constant	0.069	0.110	0.069	$(^\circ C)^{-1}$
θ for k_h	Hydrolysis constant	0.05			$(^\circ C)^{-1}$

Kinetic Parameters of Facultative Aerobic Floc-Forming Microorganisms

Depending on the solids retention time, influent and reactor configurations, about 90% of the microorganisms in the activated sludge are heterotrophic microorganisms. Therefore if the growth of nitrifiers is suppressed by a nitrification inhibitor the behavior of the biocenoses represents more or less the behavior of the floc-forming microorganisms and kinetic parameters determined for the whole biocenoses would be identical with the ones of the heterotrophic floc-forming microorganisms (Kappeler and Gujer, 1994a).

Kappeler and Gujer (1992) demonstrated that the maximum specific growth rate μ_{max} can be estimated in a batch-test by measuring the oxygen uptake rate of activated sludge, which is strongly diluted with filtered wastewater. The maximum specific growth rate μ_{max} can be calculated with equation 2.11, which is valid for biomass growth with neither substrate nor oxygen limitation:

$$\left(\frac{r_{O_2}(t)}{r_{O_2}(t_0)}\right) = e^{(\mu_{max} - \eta_{endog} b')t} \qquad 2.11$$

Where, $r_{O_2}(t)$ is oxygen uptake rate at time t (gO_2/m^3d), $r_{O_2}(t_0)$ is oxygen uptake rate at the beginning of the experiment (gO_2/m^3d), μ_{max} is maximum specific growth rate (d^{-1}), η_{endog} is correction factor for endogenous respiration which has no unit, b is coefficient for lysis (d^{-1}) and t is time variable (d) (Kappeler and Gujer, 1994a).

The yield coefficient of heterotrophic floc-forming and filamentous microorganisms, Y_H, (g COD/g COD), can be determined by mathematical optimization of respiration measurements of batch-tests under various conditions. For the heterotrophic biomass a yield coefficient of about 0.63 g COD/g COD was determined (Wanner *et al.*, 1992).

Apparent saturation coefficient K_S can be estimated by the following equation based on the simple models which consider the diffusion of soluble compounds into a reactive biological floc:

$$K_S = K_{S_o} + \left(\frac{\mu_{max} \cdot \gamma_{Floc} \cdot \delta}{Y_H \cdot a \cdot D_S} \right) \qquad 2.12$$

Where, K_{S_o} is intrinsic half-saturation coefficient for readily biodegradable substrate from the influent (g COD/m^3). The γ_{Floc} is biomass density (g/m^3 flocs), δ is diffusional length within the flocs (m), a is specific surface area of flocs (m^2/m^3) and D_S is substrate diffusion coefficient (m^2/d) (Kappeler and Gujer, 1994a).

A K_S value of about 8 g COD/m^3 for the apparent influent substrate half saturation concentration for heterotrophic floc-forming microorganisms in a typical floc diameter d of 0.6 mm was calculated based on the following assumptions: intrinsic substrate half-saturation coefficient with no additional diffusional resistance, K_{S_o} =1 g COD/m^3, the maximum specific growth rate μ_{max} (3 d^{-1}), biomass density within flocs, γ_{Floc} = 5000 g/m^3 flocs, typical diffusional length δ = 1.5x10^{-4} m, the yield coefficient Y_H = 0.63, specific surface of flocs a (6/d.1x10^4 m^{-1}) and diffusion coefficient of substrate D_S = 0.5x10^{-4} m^{-2}/d (Kappeler and Gujer, 1994a).

Apparent substrate half-saturation coefficient K_H for hydrolyzed substrate may be estimated by Equation 2.12. Consumption of the hydrolyzed products takes place where hydrolysis occurs. Therefore, the diffusional length δ is nearly zero and the K_H value is about 1 g COD/m^3 (Kappeler and Gujer, 1994a).

The oxygen half-saturation coefficient K_{O_2} for heterotrophic biomass was determined in batch tests and the nitrate half-saturation coefficient K_{NO_3} was assumed according to the recommendations of the authors of the activated sludge model No.1 as 0.25 g O_2/m^3 and 0.5 g NO_3-N/m^3 respectively. The ammonium half-saturation coefficient K_{NH_4} is determined to be 0.01 g NH_4-N/m^3 and it is expected to be smaller than the measured K_{NH_4} value for nitrifiers. This is because heterotrophic growth is generally limited by substrate and not ammonium (Kappeler and Gujer, 1994a).

The coefficient for reduced anoxic growth η_g was taken from Henze *et al.*, (1986) as 0.8. It is verified by nitrate measurements during anoxic growth in batch tests

The lysis coefficient, b', was assumed according to the authors of the activated sludge model No.1 as 0.5 d^{-1} (Henze *et al.*, 1986). Correction factor for endogenous respiration η_{endog} was assumed to be 0.4. For the determination of these values, yield coefficient, Y_H, is considered to be 0.63 g COD/g COD (Kappeler and Gujer, 1994a).

The ratio of released inert particulate COD to decayed biomass COD for endogenous respiration (f_P= 0.20 g COD/g COD) was determined. The ratio of released inert particulate COD to decaying biomass COD for lysis (f_I=0.08 g COD/g COD) is calculated according to equation 2.13:

$$f_I = f_P.(1-Y_H)/(1-f_P.Y_H) \qquad 2.13$$

Nitrogen relative to COD in all microorganisms is assumed to be 6% (i_{XB}=0.06 g N/g COD) (Kappeler and Gujer, 1994a).

Kinetic Parameters of Nitrifying Microorganisms

A big part of the kinetic parameters of the nitrifiers were determined by curve fitting the data from respiration measurements of pH-controlled batch-tests. A well known dose of ammonium-chloride is used in these tests. All curve fittings were carried out with the ASIM simulation program (Kappeler and Gujer, 1994a).

It is seen that the measured maximum specific growth rate μ_{max} is in agreement with typical values in the literature: 0.9 d^{-1} at 20°C (Kappeler and Gujer, 1994a).

For nitrification 4.33 g O_2 are consumed per g NO_3-N produced. COD-equivalent of nitrate is -4.57 g COD/g NO_3-N from the COD-continuity. Then the yield coefficient for nitrifiers, Y_A, is calculated as 4.57-4.33=0.24 g $COD_{produced}$/g $N_{oxidized}$ (Kappeler and Gujer, 1994a).

Autotrophic oxygen uptake rate was measured in order to obtain the oxygen half-saturation coefficient K_{O_2}. Depending on the experimental results it was seen that, at oxygen concentrations below 1 g O_2/m^3 a high K_{O_2} value (0.6 g O_2/m^3) should be used, whereas at 2 g O_2/m^3 a lower one (0.2 g O_2/m^3) would be better since the Monod-function is not able to describe the oxygen dependency accurately (Kappeler and Gujer, 1994a).

The ammonium half-saturation coefficient K_{NH4} was also determined by curve fitting. It was determined that 0.4 g NH_4-N/m^3 is a reasonable value for K_{NH4} (Kappeler and Gujer, 1994a).

Coefficient for autotrophic lysis, b′ and the correction factor for endogenous respiration, η_{endog}, were assumed to be 0.14 d^{-1} and 0.4 respectively and the coefficients f_I and f_P were assumed to be equal for all microorganisms (Kappeler and Gujer, 1994).

Kinetic Parameters of Obligate Aerobic Filamentous Microorganisms

Filamentous microorganisms generally represent only a small fraction of the biocenoses. Therefore, their kinetic parameters are estimated with indirect methods.

A sequencing batch reactor with activated sludge, which was strongly diluted with centrifugated wastewater, was operated for the estimation of the maximum specific growth rate of Type 021N under neither substrate nor oxygen limitation conditions. As the oxygen uptake rate decreased and the activated sludge was settled, a new portion of wastewater was added. Initial value of the rate of Type 021N concentration in the total biomass was compared with the value after 24 hours. Since the final value of the rate of Type 021N to the total biomass was similar to the initial one, it can be concluded that Type 021N and heterotrophic floc-formers have a similar maximum specific growth rate μ_{max} (Kappeler and Gujer, 1994a).

Maximum specific growth rate of *Sphaerotilus natans* is about 1.5 d^{-1} at 17°C considering that the measured μ_{max} values only decrease by about 25% for a temperature reduction from 17 to 6°C, the maximum specific growth rate of obligate aerobic filamentous microorganisms is assumed to be 3.0 d^{-1} at 20°C, similar to the floc-formers (Kappeler and Gujer, 1994a).

The yield coefficient, Y_H, was assumed to be same for facultative aerobic floc-forming and obligate aerobic filamentous microorganisms. The apparent substrate half-saturation coefficient of obligate aerobic filamentous microorganisms for readily biodegradable substrate from the influent, K_S, for hydrolysis products, K_H, and for dissolved oxygen, K_{O_2}, can be estimated from equation 2.12 as 1 g COD/m^3, 4 g COD/m^3 and 0.1 g O_2/m^3 respectively (Kappeler and Gujer, 1994a).

For the case of ammonium from the influent, Equation 2.12 could lead to the assumption that the ammonium half-saturation coefficient K_{NH4} might be lower for filamentous microorganisms than for floc-formers. But for the case of ammonium from hydrolysis it is just the opposite. Based on this, equal K_{NH4} values (0.01 g NH_4-N/m^3) for floc-formers and filaments were assumed (Kappeler and Gujer, 1994a).

The coefficient for lysis, b′, and the correction factor for endogenous respiration, η_{endog}, were supposed to have almost the same value as for floc-formers as 0.5 d^{-1} and 0.4 respectively (Kappeler and Gujer, 1994a).

Hydrolysis

The hydrolysis constant, k_h, could be estimated by interpreting the course of the oxygen uptake rate of several batch tests as 5 d^{-1} at 20°C. The reduction of the hydrolysis constant under anoxic conditions, η_h, was obtained with anoxic batch-tests as 0.6. Several measurements with typical Swiss wastewater from domestic origin showed that about 4% of the hydrolysed COD is released as NH_4-N. Therefore i_{XS}=0.04 g N/g COD was adopted for simulation (Kappeler and Gujer, 1994a).

The ratio of released soluble inert COD to hydrolysed COD, f_H, was determined by an anoxic batch-test. From the denitrification rate, the hydrolysis rate could be determined with the aid of the stoichiometry, and then it is compared with the increase of the soluble COD in the batch-test. f_H is found to be 0.07 (gCOD/gCOD) (Kappeler and Gujer, 1994a).

Wastewater Characterization

Typical wastewater fractions of domestic wastewater are listed in Table 2.7. These values were obtained from several batch-tests with wastewater from different plants (Kappeler and Gujer, 1994a).

Table 2.7 Typical Wastewater Fractions (20°C) (Kappeler and Gujer, 1994a)

Soluble Fractions			
S_{O2}	Dissolved oxygen	1	gO_2/m^3
S_S	Readily biodegradable substrate	35	$gCOD/m^3$
S_H	Hydrolysis products	0	$gCOD/m^3$
S_I	Inert COD	20	$gCOD/m^3$
S_{NH4}	Ammonium	25	gN/m^3
S_{NO3}	Nitrate	2	gN/m^3
Particulate fractions			
X_S	Slowly biodegradable substrate	200	$gCOD/m^3$
X_I	Inert organic particles	30	$gCOD/m^3$
X_{Floc}	Floc-formers	30	$gCOD/m^3$
X_{Nitr}	Nitrifiers	0.3	$gCOD/m^3$
X_{Fil}	Filaments	0.3	$gCOD/m^3$
COD_{tot}	Total COD	315	$gCOD/m^3$
$Kj\text{-}N_{tot}$	Total Kjeldahl-N	35	gN/m^3

2.8 AQUASIM for the Identification and Simulation of Aquatic Systems

Natural systems are tremendously complex and involving laboratory or technical methods make it difficult to gain insight into basic mechanisms within the system. Since the mathematical models allow exact calculation of the time evolution of a system with given interactions, they can be helpful for the analyses of complex natural systems. Mathematical models of natural systems require drastic simplifications of real processes. For this reason, they usually contain empirical model parameters to be determined with experimental data (Reichert, 1994).

In the large majority of environmental simulation programs, a given model or a strongly limited set of models is implemented. This design makes is impossible to use such programs for system identification, because the process of system

identification consists of comparing the results of different models with measured data in order to find the most appropriate model for the investigated system.

During the years 1991-1994 in the Computer and Systems Sciences department of the Swiss Federal Institute for Environmental Science and Technology (EAWAG) a more universal computer program for the identification and SIMulation of AQUAtic systems (AQUASIM) was developed. This program is applicable to a wide class of aquatic systems (Reichert *et al.*, 1995).

In this study, AQUASIM program is used as a simulation tool. Study is carried out by coding the already developed AEROFIL model in AQUASIM program and using this code for modeling activated sludge population dynamics.

CHAPTER 3

MATERIALS AND METHODS

3.1 Configuring AQUASIM Program for AEROFIL Model

AQUASIM is a program for the identification and simulation of aquatic systems. It performs the four tasks of: simulation, identifiability analysis, parameter estimation, and uncertainty analysis. Program allows the users to build their own systems. AEROFIL is a mathematical model for aerobic bulking that describes the behavior of facultative aerobic floc-forming, obligate aerobic filamentous and nitrifying microorganisms. For the purpose of modeling activated sludge population dynamics, AEROFIL model is inserted into the AQUASIM program version 2.0 (win/mfc).

3.2 Menu list of the AQUASIM Program

AQUASIM program main window includes six menu lists of: File, Edit, Calc, View, Window, and Help (Reichert, 1998a). File menu with its New, Open, Close, Save, Save As, Revert to Saved, Print Options, Print to File, About, and Exit items, performs the saving, loading printing processes of the file as it is in the Windows.

AQUASIM system files can be transferred between all supported platforms using text (ASCII) data transfer. Edit menu with its System, Variables, Processes, Compartments, Links, Numerical Parameters, Delete States items, provides the edition of these items in order to define new systems.

Calc menu with its simulation, sensitivity analysis and parameter estimation items, provides the definition and start of calculation for these purposes. View menu with its Results, Toolbar and Status Bar provides us to define the graphics of the simulation results. Window menu with its Close Dialogs item provides us to clean the monitor from the dialog boxes in order to see the graphics. Help menu provides information about the program.

3.3 Model Formulation

In the program AQUASIM, a model consists of a system of ordinary and/or partial differential equations and algebraic equations describing the behavior of a given set of important state variables of an aquatic system. The differential equations for water flow and substance transport can be selected by the choice of environmental or technical compartments connected by links. User can freely specify the source terms of these equations, which describe the effect of transformation processes. The definition of such transformation processes follows closely the notation of biochemical processes, as it is familiar to environmental scientists and engineers. Variables are used to define the processes, compartments and links. In Figure 3.1 the mutual dependencies between the four subsystems of variables, processes,

compartments and links are visualized. It is evident from the figure that the variables form the basic subsystem of the formulation of processes, compartments and links. Processes must be defined before they can be activated in compartments. Finally, links are used to connect compartments that are defined before.

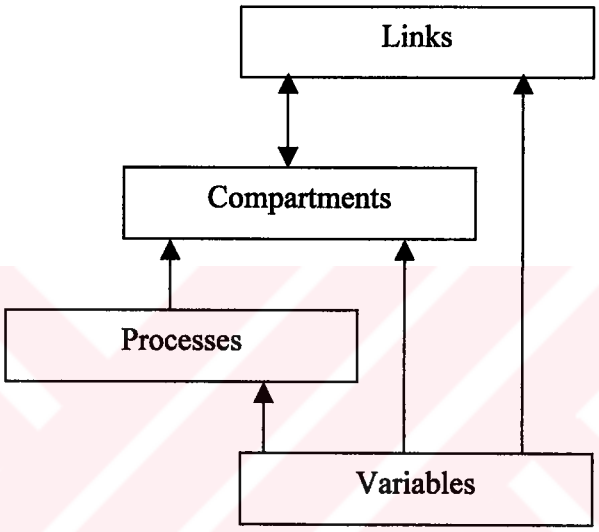

Figure 3.1 Main elements of model structure of AQUASIM

Variables

The system of **variables** forms the basic subsystem of the AQUASIM model structure. Variables are objects, which are characterized by the property of taking a possibly context-sensitive numerical value. Value of one variable may depend on the values of other variables. Six types of variables are distinguished: State variables are used to describe properties of water or of a surface in contact with water to be

calculated by the model. In activated sludge systems, state variables are used in order to describe concentrations of particulate and dissolved substances to be calculated as the solution of differential equations (Reichert *et al.*, 1995). Program variables make auxiliary quantities used in the program available to the system of variables. Constant variables and real list variables are used to provide measured quantities for use in the system of variables. In addition, constant variables are used as model parameters in sensitivity analyses and parameter estimations. Variable list variables are used to build functional relations out of other variables. Formula variables allow the user to specify algebraic expressions for describing process rates. Finally, probe variables make the values of variables evaluated at a given location in a compartment globally available. The system of variables serves as a pool of variables for the formulation of the other subsystems.

Processes

System of **processes** is the next subsystem of the AQUASIM model structure. Two types of processes are distinguished: Dynamic processes implement transformation or transfer processes, which are characterized by a common process rate and by individual stoichiometric coefficients describing the relative effect to different variables. Time evolution of variables affected by dynamic processes is determined by the solution of differential equations. The second type of processes is equilibrium process, which determines the values of the corresponding variables by the solution of algebraic equations. Equilibrium processes are used to model processes, which are so fast, that the corresponding variables can always be approximated to take their

current equilibrium values. For the formulation of biochemical process systems, as they are used for activated sludge modeling, only dynamic processes are required (Reichert *et al.*, 1995). The variables of the system of variables may be used (and are needed) to formulate processes.

Compartments

The next subsystem of the AQUASIM model structure is the system of **compartments**. This subsystem is designed to spatially divide the system under investigation. The following types of compartments are implemented in the current version of the program: Mixed reactor compartments are used to describe systems that can be approximated by an arrangement of well-mixed domains (e.g. stirred reactors, mixed lakes, etc.), biofilm reactor compartments are used to describe the growth and population dynamics of biofilms in which substrate gradients over depth are important, advective-diffusive reactor compartments can be used to describe systems with longitudinal given water flow (e.g. plug flow reactors, rivers with given water flow, etc.), saturated soil column compartments are used to model advective-dispersive transport, exchange with stagnant pore volumes, adsorption and transformation of substances in saturated soil columns, river section compartments are used to describe hydraulics, transport and transformation processes in rivers, and lake compartments are used to model stratification, mixing, transport and transformation processes in horizontally well-mixed lakes (Reichert, 1998a). Only mixed reactor compartments of fixed or variable volume are required for the formulation of the most common activated sludge systems (Reichert *et al.*, 1995).

Links

System of **links** forms the last subsystem of the AQUASIM model structure. The objects of this subsystem are to connect the compartments to the desired spatial configuration. Two types of links are distinguished to connect the compartments listed above: Advective links describe water flow and advective substance transport between compartments as well as flux separation, as it is required for selectively recirculating particulate components (Reichert *et al.*, 1995). These links cannot only directly connect compartments, but also bifurcations and junctions can be built. Diffusive links model diffusive boundary layers or membranes between compartments. These elements can be diffusively penetrated by certain substances.

3.4 Application Fundamentals of AQUASIM

Firstly, all the variables were distinguished depending on their types. Wastewater fractions S_H, S_I, S_{NH4}, S_{NO3}, S_{O2}, S_S, X_{Fil}, X_{Floc}, X_I, X_{Nitr}, and X_S are represented by dynamic volume state variables, changing concentrations of which will be computed by the program through the simulation process. Their initial values are given as stated in Table 2.7 as formula variables, which allow the user to enter context sensitive values. Kinetic parameters shown in Table 2.5 are entered as formula variables. Only μ_{max}, b′ and k_h are entered as dynamic volume state variables since their values are significantly influenced by the temperature changes. The temperature dependency formula and θ values given in the Table 2.6 are used for the computation of parameter values at different temperatures. The θ value is entered as formula

variable for each of the three parameters. Then, processes as shown in Table 2.4 are entered one by one. Since activated sludge process is best described by dynamic processes (Reichert *et al.*, 1995) all of the processes are assigned as dynamic. In order to implement the process rates, stoichiometric coefficients as stated in the Table 2.3 are used.

After the introduction of variables and processes, compartments are defined. For the simulation of activated sludge process, a mixed reactor compartment and a clarifier is edited. Completely mixed reactor accepts all the inflow, Q_{in}, it has a constant volume, and all the state variables and processes are activated in this compartment. Completely mixed reactor is connected to the clarifier by an advective link. This link has one bifurcation carrying the recirculation flow, Q_{rec}, to the head of the completely mixed reactor. Variables can be edited in the recirculation flow.

Simulation is performed for different step sizes and number of steps. After some trials 0.01 step size with 10000 steps (simulating 100 days of operation) has been found to be suitable in order to reach the steady state conditions.

Simulation results are visualized by plots. Plot definitions that will provide the concentrations of substrates and of microorganisms in each reactor are also made. By the help of these plots $X_{Fil}/X_{Biomass}$ rates are determined.

All of the variables each with assigned variable types to them are tabulated in table 3.1 below.

Table 3.1 Variable types according to AQUASIM program

Symbol	Variable name	Variable type
b'	Coefficient for lysis (d^{-1})	Formula variable b'(20°C)*exp(θb'*(T-20))
$b'(20^oC)$	Coefficient for lysis at 20°C (d^{-1})	Formula variable
f_H	Ratio of released soluble inert COD to hydrolysed COD (gCOD/gCOD)	Formula variable
f_I	Ratio of released inert particulate COD to decayed biomass COD for lysis (gCOD/gCOD)	Formula variable
f_P	Ratio of released inert particulate COD to decayed biomass COD for endogenous respiration (gCOD/gCOD)	Formula variable
η_{endog}	Correction factor for endogenous respiration (-)	Formula variable
η_g	Coefficient for reduced anoxic growth (-)	Formula variable
η_h	Coefficient for reduced anoxic hydrolysis (-)	Formula variable
i_{XB}	Ratio of nitrogen to COD in biomass (gN/gCOD)	Formula variable
i_{XS}	Ratio of ammonium-nitrogen to COD released by hydrolysis(gNH_4-N/gCOD)	Formula variable
k_h	Hydrolysis constant (d^{-1})	Formula variable k_h(20°C)*exp(θk_h*(T-20))
$k_h(20^oC)$	Hydrolysis constant at 20°C (d^{-1})	Formula variable
K_H	Half-saturation coefficient for hydrolysis products ($gCOD/m^3$)	Formula variable
K_{NH4}	Half-saturation coefficient for ammonium-nitrogen (gNH_4-N/m^3)	Formula variable
K_{NO3}	Half-saturation coefficient for nitrate-nitrogen (gNO_3-N/m^3)	Formula variable
K_{O2}	Half-saturation coefficient for dissolved oxygen (gO_2/m^3)	Formula variable
K_S	Apparent half-saturation coefficient for readily biodegradable substrate from the influent ($gCOD/m^3$)	Formula variable
K_{So}	Intrinsic half-saturation coefficient for readily biodegradable substrate from the influent ($gCOD/m^3$)	Formula variable
KXO_2	Aeration coefficient	Constant variable
$MLSS$	Mixed liquor suspended solids ($gCOD/m^3$)	Formula variable (Xfil+Xfloc+Xnitr+XI+XS)/0.8
μ_{max}	Maximum specific growth rate (d^{-1})	Formula variable μ_{max}(20°C)*exp(θμ_{max}*(T-20))

Table 3.1 (continued) Variable types according to AQUASIM program

Symbol	Variable name	Variable type
$\mu_{max}(20^{\circ}C)$	Maximum specific growth rate at 20°C	Formula variable
Q	Discharge (m^3/d)	Program variable with reference to "discharge"
Q_{in}	Influent flow rate (m^3/d)	Formula variable
Q_{rec}	Recirculation flow rate (m^3/d)	Formula variable
SRT	Solids retention time (d)	Formula variable
S	Soluble matter ($gCOD/m^3$)	Dynamic volume state variable
S_{in}	Inflow concentration of soluble matter ($gCOD/m^3$)	Formula variable
S_H	Hydrolysis products ($gCOD/m^3$)	Dynamic volume state variable
S_I	Soluble inert COD ($gCOD/m^3$)	Dynamic volume state variable
S_{NH4}	Ammonium-nitrogen ($gNH_4\text{-}N/m^3$)	Dynamic volume state variable
S_{NO3}	Nitrate-nitrogen ($gNO_3\text{-}N/m^3$)	Dynamic volume state variable
S_{O2}	Dissolved oxygen (gO_2/m^3)	Dynamic volume state variable
S_S	Readily biodegradable substrate from the influent ($gCOD/m^3$)	Dynamic volume state variable
T	Temperature (°C)	Formula variable
t	Time variable (d)	Program variable with reference to "time"
θ	Coefficient for temperature dependence $(^{\circ}C)^{-1}$	Formula variable
X	Particulate matter ($gCOD/m^3$)	Dynamic volume state variable
X_{in}	Inflow concentration of particulate matter ($gCOD/m^3$)	Formula variable
X_{Fil}	Obligate aerobic filamentous m.o. ($gCOD/m^3$)	Dynamic volume state variable
X_{Floc}	Facultative aerobic floc-forming m.o. ($gCOD/m^3$)	Dynamic volume state variable
X_I	Inert suspended organic matter ($gCOD/m^3$)	Dynamic volume state variable
X_{Nitr}	Nitrifying microorganisms ($gCOD/m^3$)	Dynamic volume state variable
X_S	Slowly biodegradable substrate ($gCOD/m^3$)	Dynamic volume state variable
$X_{Biomass}$	Total biomass in the activated sludge ($gCOD/m^3$)	Formula variable ($X_{Floc}+X_{Fil}+X_{Nitr}$)
V	Reactor volume (m^3)	Program variable with reference to "reactor volume"
Y_A	Yield coefficient of autotrophic m.o. (gCOD/gN)	Formula variable
Y_H	Yield coefficient of heterotrophic floc-forming and filamentous m.o. (gCOD/gCOD)	Formula variable

CHAPTER 4

RESULTS AND DISCUSSION

4.1 Description of the Computer Model

The AEROFIL model was developed by Kappeler and Gujer (1994b) in order to describe the competition between facultative aerobic floc-forming microorganisms, obligate AERObic FILamentous microorganisms and nitrifying microorganisms in activated sludge culture. They used the typical bio-kinetic parameters and wastewater fractionation in designing the model, and have attempted to predict aerobic bulking conditions with a computer program. The model was once verified by the use of the ASIM (Gujer, 1990) simulation program by Kappeler and Gujer (1994b).

In the present study it was attempted to re-simulate floc formation by writing the already developed AEROFIL model into the AQUASIM utility software and developing a computer code for the purpose of modeling activated sludge population dynamics.

The AEROFIL model was slightly modified in this study to exclude oxygen in some of the tanks to accommodate anoxic growth in these reactors. The modifications undertaken are discussed in the relevant context.

After coding AEROFIL model in AQUASIM program, the simulation was tested by using the literature data. Upon verification of the computer model by using the literature data, several trial runs were made in order to predict the effects of several parameters on the floc characteristics and to support the model.

Trial runs were undertaken to show the effects of the following on activated sludge biocenoses:

Prediction of the Effects of Various Parameters on Sludge Floc Composition by Using AQUASIM

1. High loadings with readily biodegradable substrate,
2. Influent pattern into longish aeration basins,
3. Inoculation of facultative aerobic floc-former microorganisms,
4. Inoculation of obligate aerobic filamentous microorganisms,
5. Aerobic selectors,
6. Selector compartmentalization,
7. Anoxic selectors,
8. Anaerobic selectors

All these effects were investigated by using the AEROFIL in AQUASIM.

Moreover, effects of several biological nutrient removal (BNR) activated sludge modifications on floc biocenoses were investigated in the following configurations by using AEROFIL in AQUASIM:

Prediction of Sludge Floc Composition in Advanced BNR Systems by Using AQUASIM

1. Completely mixed activated sludge,
2. The A/O process,
3. The A^2/O process,
4. Bardenpho process,

The AEROFIL model was useful in predicting the concentrations of facultative aerobic floc-forming as well as obligate aerobic filamentous and nitrifying microorganisms in the activated sludge in COD units. Without such a tool it is exceedingly difficult to determine the relative abundance of the individual groups.

The verification and model supporting study of the AEROFIL model was carried out by Kappeler and Gujer, 1994b with an experiment on two pilot plants in the wastewater treatment plant of Nänikon and by a full-scale experiment in the wastewater treatment plant of Dagmarsellen in Switzerland. The reactor configurations and wastewater fractions, used in these experiments are listed through examples 1-7 in Table 4.1. For the BNR configurations examples 8-10 in Table 4.1 are used.

Table 4.1 Reactor configurations and wastewater fractions (Kappeler and Gujer, 1994b)

Simulation	Nänikon, CM (apple juice)	Longish aeration basins	Primary settling tanks	Inoculation of filaments	Aerobic selectors	Compartments aerobic selectors	Anoxic selectors	A/O	A^2/O	Bardenpho (4 stage)
Examples	Example 1	Example 2	Example 3	Example 4	Example 5	Example 6	Example 7	Example 8	Example 9	Example 10
Reactor configuration										
Volume (m^3)	0.25	4000 (V_{tot})	4000	4000	4000(V_{tot})	400+3600	4000(V_{tot})	4000(V_{tot})	5000(V_{tot})	3000(V_{tot})
O_2 (gO_2/m^3)	2	2	2	2	2	2	0/2	0/2	0/0/2	0/2/0/2
Q_{in} (m^3/d)	1.8	12000	12000	12000	12000	12000	12000	10000	10000	10000
Q_{Rec} (m^3/d)	1.8	12000	12000	12000	12000	12000	12000	15000	15000	15000
SRT (d)	3	6	6	6	2/6/10	2/6/10	2/6/10	3	15	15
Temp.(°C)	16	20	20	20	10/20	20	10/20	15	15	15
Wastewater										
COD_{tot} ($gCOD/m^3$)	200/260	315	195-435	290-330	315/330	315/330	315/330	315	315	315
Kj-N (gN/m^3)	21/22	35	30-40	35	35	35	35	35	35	35
S_{O2} (gO_2/m^3)	1	1	1	1	1	1	1	1	1	1
S_S ($gCOD/m^3$)	10/50	35	35	10-50	35/50	35/50	35/50	35	35	35
S_H ($gCOD/m^3$)	0	0	0	0	0	0	0	0	0	0
S_I ($gCOD/m^3$)	20	20	20	20	20	20	20	20	20	20
S_{NH4} (gN/m^3)	15	25	25	25	25	25	25	25	25	25
S_{NO3} (gN/m^3)	2	2	2	2	2	2	2	2	2	2
X_S ($gCOD/m^3$)	130/150	200	100-300	200	200	200	200	200	200	200
X_I ($gCOD/m^3$)	20	30	30	30	30	30	30	30	30	30
X_{Floc} ($gCOD/m^3$)	20	30	10-50	30	30	30	30	30	30	30
X_{Nitr} ($gCOD/m^3$)	0.2	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3
X_{Fil} ($gCOD/m^3$)	0.2	0.3	0.3	0-4	0.3	0.3	0.3	0.3	0.3	0.3

4.2 Sludge Volume Index Calibration Curve

It was already demonstrated (Sezgin *et al.*, 1980; Palm *et al.*, 1980; Walker, 1982; Lee *et al.*, 1983; Matsui and Yamamoto, 1984; Nowak *et al.*, 1986) that a direct relationship exists between the length of the filamentous microorganisms and the sludge volume index (SVI) in activated sludge. Therefore, model predictions of the SVI were carried out by estimating the concentration ratio of filamentous microorganisms to total biomass ($X_{Fil}/X_{Biomass}$) by the simulation model and then by consulting to the calibration curve presented in Fig. 4.1, as prepared by Kappeler and Gujer (1994b) using actual data. Here, total biomass, $X_{Biomass}$, refers to the sum of aerobic filamentous microorganisms, facultative aerobic floc-forming microorganisms and nitrifying microorganisms.

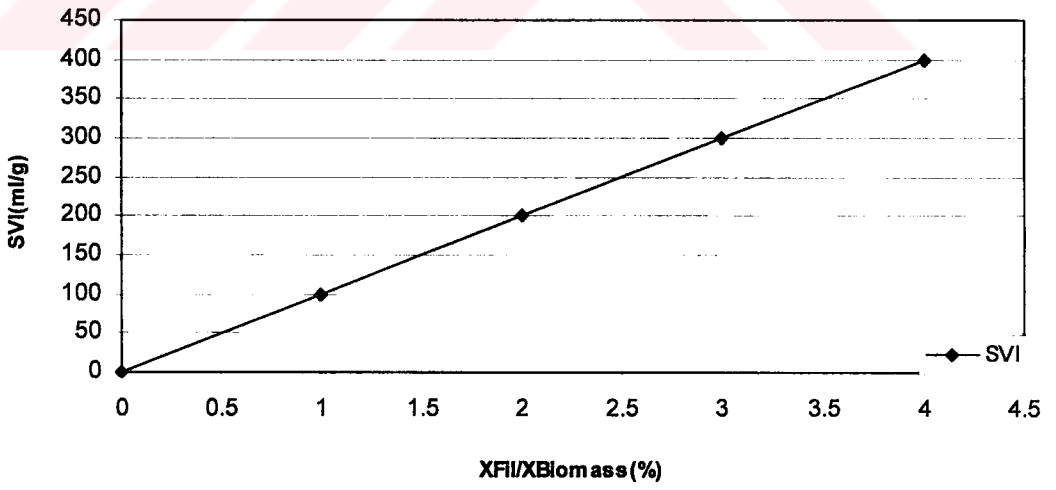

Figure 4.1 Sludge volume index (SVI) to $X_{Fil}/X_{Biomass}$ calibration curve (Kappeler and Gujer, 1994b)

4.3 High Loadings With Readily Biodegradable Substrate

In the Nänikon wastewater treatment plant, one pilot plant with a completely mixed reactor, CM, was operated at 3 days solids retention time and 16°C during March-April 1990. Reactor characteristics and wastewater fractions used for the simulation are presented in example 1 in Table 4.1. The amount of readily biodegradable substrate in the influent was very low until day 35. The primary effluent used for experiments was of domestic origin. COD_{tot} was about 200 g COD/m^3 and readily biodegradable substrate, S_{So}, was about 10 g COD/m^3. Since the fraction of readily biodegradable substrate in the total COD was small, bulking did not occur. In order to provoke aerobic bulking, about 60 g COD/m^3 were added to the primary effluent in the form of apple juice after day 35. Addition of the apple juice led to a proliferation of Type 021N filaments in the completely mixed reactor CM. The sludge volume index increased from 100 to 400 ml/g within 1 week.

For the simulation of the case with AEROFIL in AQUASIM, a mixed reactor compartment was edited. By the help of variable list variables with time reference, after day 35, influent readily biodegradable substrate concentration was increased to 60 g COD/m^3 from 10 g COD/m^3. Simulation was performed for 42 days.

Predicted sludge volume indices of the AEROFIL model using ASIM program by *Kappeler and Gujer, (1994b)*, predicted sludge volume indices of AEROFIL model using AQUASIM program *in this study* and the *measured* sludge volume indices are presented collectively in Figure 4.2. As can be seen from this figure, the predictions by AQUASIM were slightly better than that by ASIM.

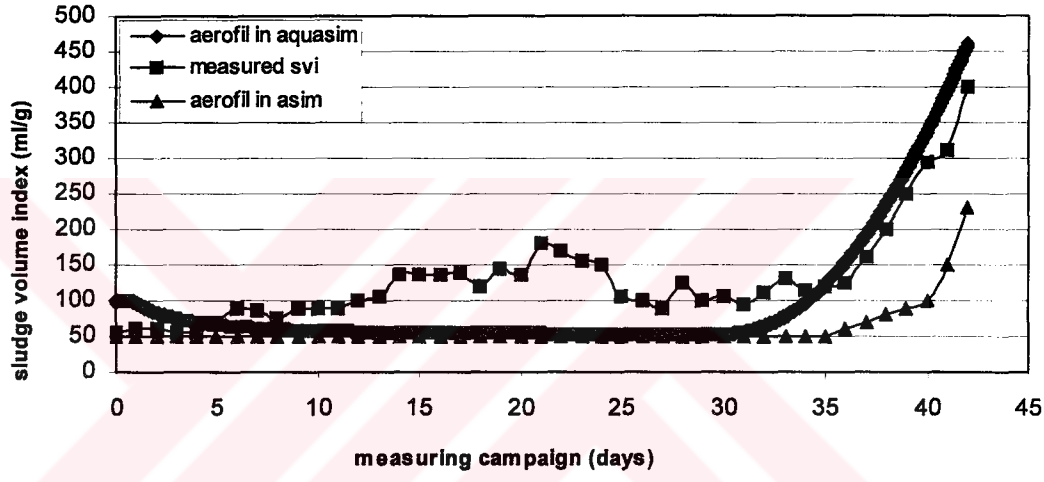

Figure 4.2 High loading with readily biodegradable substrate to a completely mixed reactor. Comparison of measured SVIs with the SVIs predicted by AEROFIL in AQUASIM and AEROFIL in ASIM.

4.4 Influent Pattern Into Longish Aeration Basins

It is known from the practice that upon changing the influent pattern into longish aeration basins from step feed to head end feed, settling properties of activated

sludge improve. It is because this modification leads to a change in flow characteristics from completely mixed towards plug flow, resulting in higher concentrations of readily biodegradable substrate in the inlet zone of the reactor and it favors the growth of floc-forming microorganisms.

For the simulation of the effect of changing influent pattern into longish aeration basins on the activated sludge biocenoses plant characteristics and wastewater composition in example 2 in Table 4.1 are used. Longish aeration basin with step feed is simulated as a completely mixed reactor whereas head end feed is simulated as a cascade of three to four identical completely mixed reactors.

The predictions of AEROFIL model with AQUASIM program is in excellent agreement with the predictions of AEROFIL model with ASIM program. The predictions of the AEROFIL model are presented in Figure 4.3. The prognoses for the fraction of obligate aerobic filamentous microorganisms correspond to the observations that head end feed (n=3 or 4) reduced the amount of filaments, but as it can be seen from the figure that settling properties may be improved by a selector system (n=10).

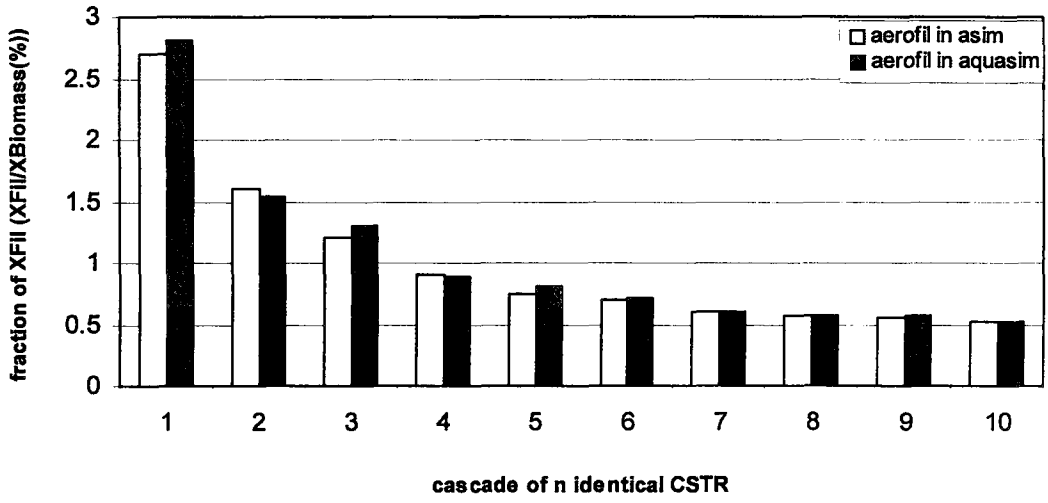

Figure 4.3 Influence of flow characteristics on the fraction of filamentous microorganisms in the activated sludge as predicted by AEROFIL in ASIM and by AEROFIL in AQUASIM

From Figure 4.3 it can be seen that compartmentalization decreases the $X_{Fil}/X_{Biomass}$ ratio by changing the reactor characteristics from completely mixed to plug flow, similar to selector effect. Therefore, with reference to Figure 4.1, it can be stated that SVI of the system should also decrease. This result is in agreement with the study of Gabb *et al.*, 1996. Unfortunately no real data was available to test the validity of the program in this case, however the results agree well with the literature. For example, Chambers and Tomlinson (1982) documented the effect of compartmentalization in terms of SVI by data from 24 activated sludge plants in England. Jenkins (1984) also reported the same effect by using the identical data but expressed hydraulics in terms of number of equivalent tanks in series. Their analyses are presented in Figure 4.4.

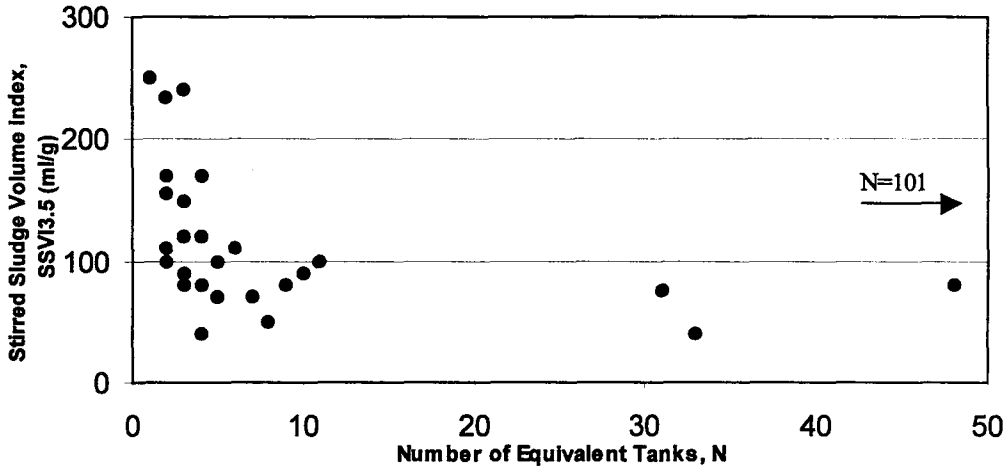

Figure 4.4 Effect of compartmentalization on SSVI in 24 English Plants (Tomlinson, 1982)

4.5 Inoculation of Facultative Aerobic Floc-forming Microorganisms

Hydraulic retention time in the primary settling tanks strongly affects the concentrations of particles in the primary effluent, especially the concentrations of slowly biodegradable substrates and the incoming heterotrophic floc-formers to the aeration tank (Kappeler and Gujer, 1994b). Aerobic bulking is a competition phenomenon between floc-forming and filamentous microorganisms. Thus, everything that increases floc-forming biomass causes reduced filamentous growth at the same time. Hence, if the particulate COD fraction of the influent is reduced, fraction of filaments in the activated sludge increases. Large primary clarifiers reduce the particulate COD fractions, thus the aerobic bulking is favored.

For the simulation of this effect, slowly biodegradable substrate, X_{S0}, and inoculated heterotrophic floc-former, $X_{Floc,0}$, concentrations were altered in the influent using the data in example 3 in Table 4.1 for a completely mixed activated sludge plant. The $X_{Fil}/X_{Biomass}$ predictions of the AEROFIL using AQUASIM in this study with respect to the slowly biodegradable substrate, X_{S0}, and inoculated heterotrophic floc-former, $X_{Floc,0}$, concentrations are summarized in Figure 4.5. From the figure it can be seen that as the floc former concentration and slowly biodegradable substrate concentration are decreased, fraction of filamentous microorganisms increased, which is a case detrimental to the sludge settling properties. Predictions by the ASIM program was identical to AEROFIL in AQUASIM, hence former was not displayed here.

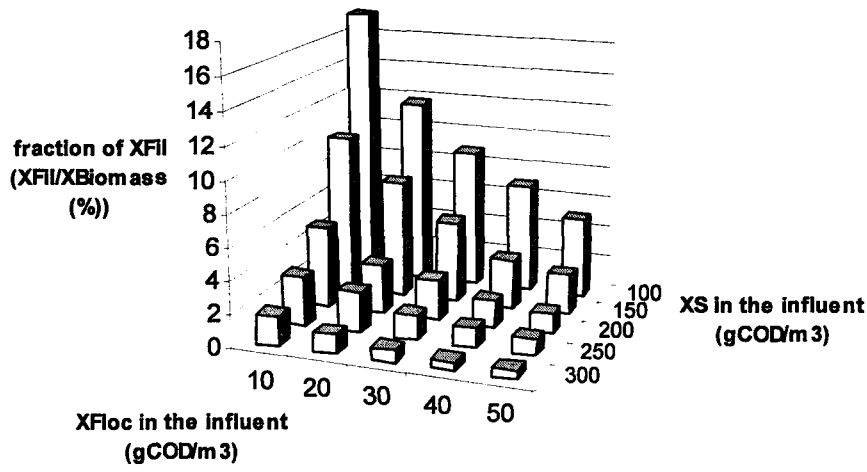

Figure 4.5 Effect of particulate slowly biodegradable substrate X_{S0} and inoculated heterotrophic floc-formers $X_{Floc,0}$ in the influent on the fraction of obligate aerobic filamentous microorganisms X_{Fil} in the activated sludge; as predicted by AEROFIL in AQUASIM

4.6 Inoculation of Obligate Aerobic Filamentous Microorganisms

Slime layers in sewers contain floc-forming as well as some filamentous microorganisms. In Switzerland *Sphaerotilus natans* and traces of 021N, *Thiothrix*, *Beggiatoa* and others are often observed in slime layers (Kappeler and Gujer, 1994b). Erosion from these slime layers leads to a continuous inoculation of biological treatment steps with these filamentous microorganisms. Concurrently, inoculation of these filamentous microorganisms causes bulking of activated sludge (Ekama *et al.*, 1996a) and as predicted in Fig. 4.5.

For the simulation of this effect, readily biodegradable substrate in the influent S_{S0} and inoculation of obligate aerobic filamentous microorganisms $X_{Fil,0}$ concentrations were altered in the influent using the data in example 4 in Table 4.1 for a completely mixed activated sludge.

Figure 4.6 shows the influence of the readily biodegradable substrate, S_{S0}, and inoculated obligate aerobic filamentous microorganisms, $X_{Fil,0}$, concentrations on the $X_{Fil}/X_{Biomass}$ ratio as predicted by the AEROFIL using AQUASIM. As the filamentous microorganism concentration increased and readily biodegradable substrate concentration increased, fraction of filamentous microorganisms was also found to increase, which is a case detrimental to the sludge settling properties.

It is seen from the figure that even if only 1% of the incoming COD consists of filamentous organisms, severe bulking in the plant may occur. If increased amounts of readily biodegradable substrate are to be present in the influent, the accumulation of these filaments in the activated sludge was especially severe.

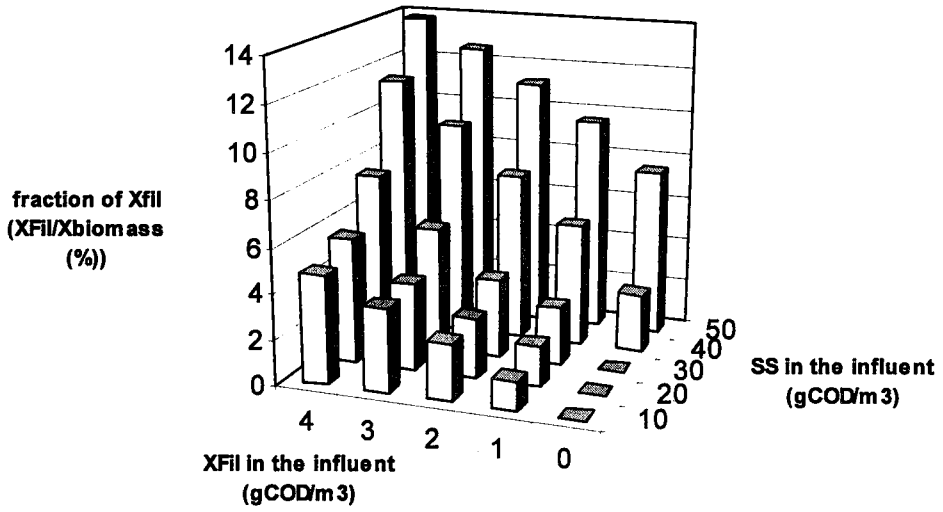

Figure 4.6 Effect of readily biodegradable substrate S_{S0} and inoculation of filamentous microorganisms $X_{Fil,0}$ in the influent on the fraction of obligate aerobic filamentous microorganisms X_{Fil} in the activated sludge; as predicted by AEROFIL in AQUASIM .

4.7 Design of Aerobic Selectors

Selectors are used in order to support floc-former microorganism growth while suppressing the filamentous microorganism growth with a substrate gradient. Flow characteristics has an influence on the settling properties of activated sludge. Chudoba and co-workers proposed to construct aerobic selectors in order to cure aerobic bulking problems (Chudoba et al., 1973a, b). With the AEROFIL model the influence of the selector volume on filamentous growth is calculated by using AQUASIM in this study for different cases of plant characteristics and wastewater

compositions. Example 5 in Table 4.1 was taken as basis. All calculations were carried out for steady-state conditions with the assumption that the concentration of dissolved oxygen is 2 g O_2/m^3 in the entire aeration basin.

The influence of selector volume on the composition of filamentous microorganisms, as indicated by a ratio between total biomass and obligate aerobic filamentous microorganisms as predicted by AEROFIL in AQUASIM is presented in Figure 4.7. If only a small amount of readily biodegradable substrate is present in the influent the amount of filaments in the biocenoses generally remains low, therefore the effect of aerobic selectors on bulking is not very significant.

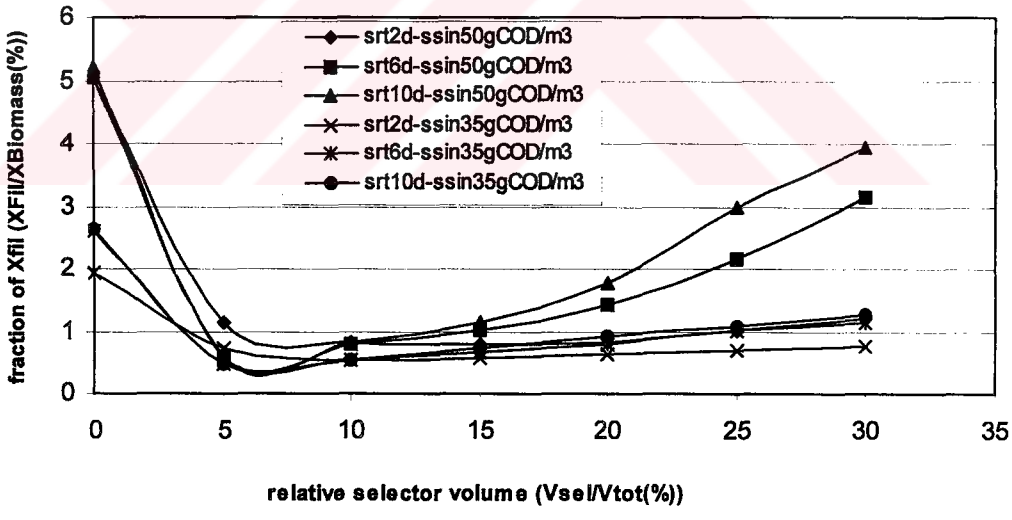

Figure 4.7 Predicted influence of the relative selector volume on the fraction of filamentous microorganisms at 20°C as predicted by AEROFIL in AQUASIM

It follows from Figure 4.7 that an optimal selector volume exists which differs little from case to case. It can be seen from the same figure that the optimum selector volume was 10-15 % of the total volume with the assumption that the optimal relative size of selectors is in the range where the predicted filaments do not exceed the minimal fraction by more than 0.5%. With decreasing temperature and solids retention time optimal selector volume should increase. This is quite reasonable since most of the readily biodegradable substrate from the influent must be consumed within the selector in order to prevent aerobic bulking. Activity of the floc-formers in the system decreases with the decreasing temperature and solids retention time, which in turn necessitates the increase of selector volume. However it can be deduced from Fig. 4.6 that, in the case of large amounts of inoculated obligate aerobic filaments, aerobic selectors may not perform all too well.

Accepting that the optimum relative selector volume is 10%, it is now possible to estimate a range for the optimal loading of the activated sludge into the selector. The calculated optimal sludge loadings into the aerobic selectors are presented in Table 4.2 for different cases as indicated in example 5 in Table 4.1. Calculations are made for a usual BOD_5/COD-ratio of primary effluent of 0.5 g BOD_5/g COD. In order to calculate the optimal sludge loadings Equation 4.1 presented below was used.

$$\frac{F}{M} = \left[\frac{BOD_5 * Q_{in}}{X_S * V_S} \right] \qquad 4.1$$

Where,

F/M= Food to Microorganism ratio

BOD_5= Concentration in kg BOD_5/m^3

Q_{in}= Influent flow rate (m^3/d)

X_S= MLSS concentration in the selector

V_S= Selector volume (m^3)

The calculation of the optimal selector sludge loadings in Table 4.2 were carried out according to the following protocol:

The BOD_5 concentration in above calculations was converted from COD output of the model. In example 5, which was used to verify the AQUASIM model, the total COD was equal to 315 g COD/m^3 and the influent readily biodegradable substrate was 35 g COD/m^3. In the same example when total influent COD was 330 g COD/m^3 influent readily biodegradable substrate was 50 g COD/m^3. These values were converted to BOD units by using the typical BOD_5/COD-ratio of 0.5 applicable for primary effluents. Thus, influent BOD_5 concentration works out as 0.1575 kg BOD/m^3 and 0.165 kg BOD/m^3, respectively. The influent flow rate, Q_{in}, was 12000m^3/d. Since optimal relative selector volume was found to be 10% from Fig. 4.6, the selector volume was calculated as 400 m^3 for the particular example. The MLSS concentration in the selector was then determined by the AQUASIM program for each SRT (2d, 6d, 10d), SS_{in} values (35 gCOD/m^3, 50gCOD/m^3) and for temperatures of 10°C and 20°C. The calculated F/M values are presented in Table 4.2.

Table 4.2 The calculated aerobic optimal selector sludge loadings for 10% relative selector volume

Aerobic selector sludge loading (kg BOD_5/kg MLSS d)		
SS_{IN}=35 gCOD/m^3	T=10°C	T=20°C
SRT2	3.3	3.5
SRT6	1.59	1.84
SRT10	1.21	1.43
SS_{IN}=50 gCOD/m^3	T=10°C	T=20°C
SRT2	3.36	3.6
SRT6	1.62	1.89
SRT10	1.23	1.46

In Germany the ATV Task Group on Bulking (1988) suggested that aerobic selectors should be designed with a volumetric loading of 10 kg BOD_5/m^3d. Considering a typical activated sludge concentration of 3 kg MLSS/m^3 this is equivalent to a selector sludge loading of 3.3 kg BOD_5/kg MLSS d. This empirically determined value is compared with the theoretical predictions by AQUASIM in Table 4.2. Experience and theoretical predictions are found to be in good agreement.

4.8 Effect of Selector Compartmentalization on Aerobic Bulking

The ATV Task Group on Bulking also recommended to compartmentalize aerobic selectors. The effect of selector compartmentalization was studied by using data from the above-defined case of example 6 in Table 4.1. The total selector volume was selected as 10% of the total aeration volume, based on Fig. 4.7. The influence of numbers of selector compartments on the aerobic filamentous organisms /total biomass as predicted by AEROFIL in AQUASIM is presented in Figure 4.8. From

the figure it can be seen that aerobic selectors with two or three compartments can further suppress the growth of filamentous microorganisms. However more than three selector compartments do not further improve settling properties. Compartmentalization of aerobic selectors become very important with increased amounts of readily biodegradable substrate in the influent. Nitrifiers are not influenced by aerobic selectors as long as the oxygen supply is sufficient.

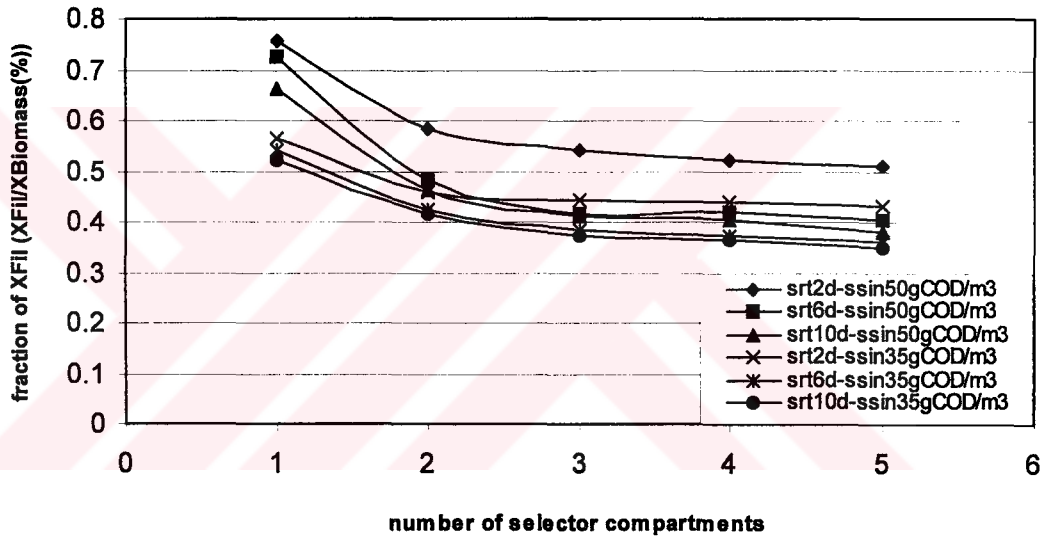

Figure 4.8 Predicted influence of compartmentalized aerobic selectors on filamentous growth as predicted by AEROFIL in AQUASIM

4.9 Design of Anoxic Selectors

Biocenoses composition can also be predicted for anoxic selectors by using AEROFIL model in AQUASIM program. For this experiment, plant characteristics

and wastewater composition in example 7 in Table 4.1 were used. The influence of anoxic selector volumes on the *obligate aerobic filamentous microorganisms / total biomass* is presented in Figure 4.9 for different operational conditions. It can be seen from the figure that anoxic selector volumes of 5% and above are effective in suppressing filamentous bulking as filamentous microorganisms are obligate aerobic.

As it were in the case in the aerobic selectors, volume of anoxic selectors should also be enlarged for good settling as temperature and SRT decreases. However this was not justifiable in Fig. 4.9. Since anoxic selectors will induce decay of filaments, this will lead to a decrease in inoculated obligate aerobic filaments into the aerobic zone.

The important role of nitrifiers in establishing anoxic selectors is apparent, since without the activity of these bacteria nitrate would not be produced in the system and denitrification conditions would not be set up. If the solids retention time is too short at low temperatures, the anoxic growth of heterotrophic floc-formers is limited by nitrate, which should serve as electron acceptor. Hence, the removal of readily biodegradable substrate from the influent is not complete within the "anoxic" selector. This causes the increased concentrations of obligate aerobic filamentous microorganisms, which means that anoxic selectors only function if the elimination of the readily biodegradable substrate from the influent is complete and not limited by nitrate. Therefore presence of sufficient nitrate is very important for well functioning anoxic selectors.

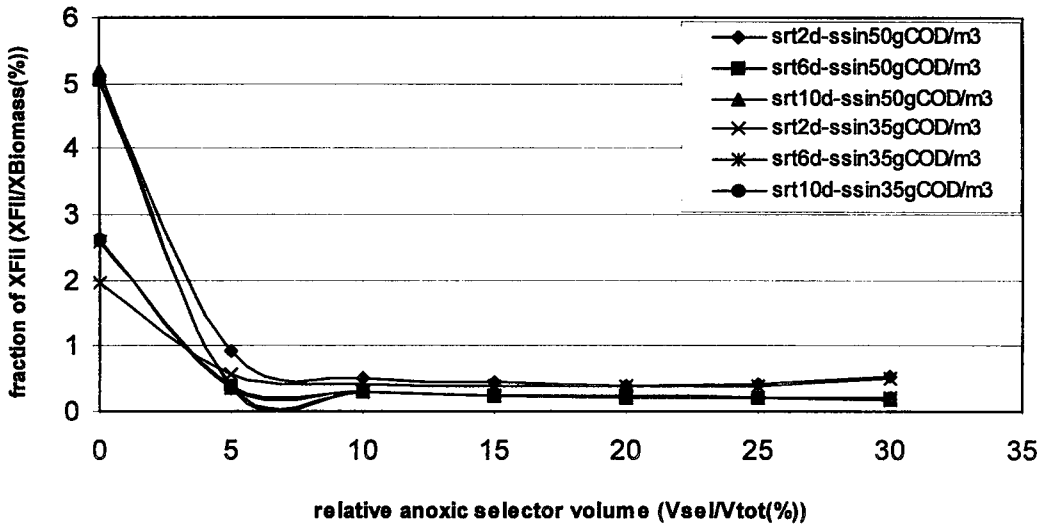

Figure 4.9 Influence of the relative volume of anoxic selectors on the fraction of filamentous microorganisms at 20°C as predicted by AEROFIL in AQUASIM

As calculated for aerobic selectors, optimal F/M ratios for the design of anoxic selectors were calculated and presented in Table 4.3. In these calculations F/M ratios leading to *filamentous microorganism / total biomass* ratio of 0.5 % or below were aimed.

The ATV standard of 2.5 kg BOD_5/m^3d or 0.8 kg BOD_5/kg MLSS d were met at 20 % anoxic selector volume, which also corresponds to *filamentous microorganism / total biomass* ratio of 0.5%. A 20 % anoxic selector volume is greater than the 10% volume determined for the aerobic selector. This is true because anoxic selectors must be sized in order to provide the utilization of all the influent readily biodegradable COD. But readily biodegradable COD uptake rate is lower under anoxic conditions. Therefore, for the same design situation, the size of the anoxic

selector should be greater than its aerobic counterpart. This was also shown by Ekama *et al.* (1996b). It is then possible to estimate a range for optimal selector loading. Calculations are carried by using Equation 4.1. Considering the influent flow rate, Q_{in}, as 12000m^3/d, and taking the relative selector volume as 20%, a selector volume of 800 m^3 was computed. The MLSS concentration in the selector was determined by AQUASIM for each SRT (2d, 6d, 10d) and SS_{in} value (35 gCOD/m^3, 50gCOD/m^3) and temperatures of 10°C and 20°C. The determined F/M values are presented in Table 4.3 for different operational conditions in example 7 in Table 4.1.

Table 4.3 The calculated Anoxic selector sludge loadings for 20% relative selector volume

Anoxic selector sludge loading (kg BOD_5/kg MLSS d)		
SS_{IN}=35 gCOD/m^3	T=10°C	T=20°C
SRT2	1.73	1.89
SRT6	0.84	0.95
SRT10	0.64	0.76
SS_{IN}=50 gCOD/m^3	T=10°C	T=20°C
SRT2	1.75	1.91
SRT6	0.82	0.93
SRT10	0.62	0.74

The ATV Task Group on Bulking recommends the following loading criteria: 2.5 kg BOD_5/m^3d or 0.8 kg BOD_5/kgMLSSd for anoxic selectors. Low temperatures are generally critical for anoxic selectors and should be considered during design. If the ATV-value of 2.5 kg BOD_5/m^3d or 0.8 kg BOD_5/kg MLSS d, would have been used no excessive growth of obligate aerobic filamentous microorganisms is to be

expected. If suppression of excessive growth of obligate aerobic filamentous microorganisms is the only goal of the denitrification zone and nitrogen elimination is not aimed, then anoxic selectors may be designed with higher sludge loadings. For typical Swiss wastewater and for temperatures above 10°C, the design sludge loading for anoxic selectors might be increased to 2.0 kg BOD_5/kg MLSS d, as recommended by Kappeler and Gujer (1994b). Somewhat higher loading rates with the lower SRTs found in Table 4.3, is quite natural as F/M ratio is inversely proportional with the SRT in the fundamental activated sludge kinetics.

The difference in the design sludge loadings for aerobic and anoxic selectors is mainly based on the fact that growth on readily biodegradable substrate under anoxic conditions is slower than under aerobic conditions. The reduction in metabolic rates of heterotrophs under anoxic growth conditions is in fact represented by a η_g factor, which is smaller than 1.0, in ASM1 model. Hence, anoxic selectors may be designed according to aerobic selector design criteria but overall volume is to be reduced by a factor of η_g.

Compartmentalization of anoxic selectors is not compulsory since obligate aerobic filamentous microorganisms are unable to grow under anoxic conditions.

4.10 Design of Anaerobic Selectors

Anaerobic selectors may be useful in reducing bulking problems. This effect is evidently due to the ability of floc-forming organisms to store the soluble COD within cells as observed in the context of biological phosphorus removal. The filamentous organisms are unable to store the soluble COD under the same conditions, hence are competed out. The AEROFIL model does not include this phenomenon.

Prediction of Sludge Floc Composition in Advanced BNR Systems by Using AQUASIM

4.11 Predicting Population Distribution in a Completely Mixed Activated Sludge Reactor

To serve as basis for comparison with the BNR systems, simulation of floc biocenoses and substrate removal in a completely mixed activated sludge process, as shown in Figure 2.1, was carried out by AEROFIL model using AQUASIM. Data from example 8, in Table 4.1 is used with some alterations. Reactor volume was altered to 1000 m^3 and oxygen concentration in the whole aeration basin was taken as 2 g COD/m^3 but influent composition was identical to the example 8. An SRT of 15 days was chosen. The soluble carbonaceous substrate concentration versus time in the aeration tank as predicted by AEROFIL in AQUASIM is shown in Figure

4.10. As can be seen from this figure S_I increases to 35 g COD/m^3 after initial start-up while S_H and S_S both approach to zero.

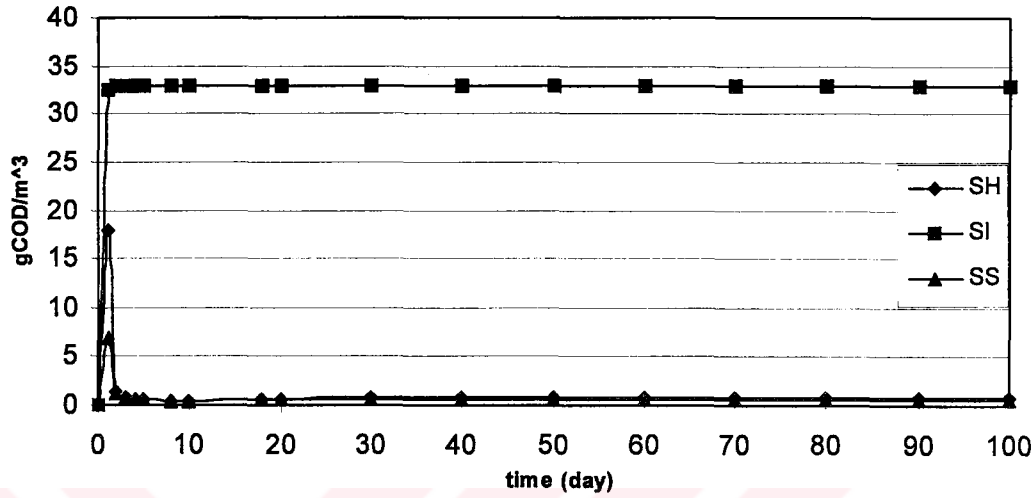

Figure 4.10 The S_H, S_I and S_S concentrations in the completely mixed aerobic reactor as predicted by AEROFIL in AQUASIM

The Figure 4.11 depicts ammonium, S_{NH4}, and nitrate-nitrogen, S_{NO3}, concentrations in the aerobic tank. As can be seen from this figure complete nitrification occurs at this SRT and effluent is thus fully nitrified. The Figure 4.12 shows the biocenoses composition in the aeration tank. $X_{Fil}/X_{Biomass}$ (%) ratio was calculated as 1.501. As can be seen from this figure, floc formers are excessively dominant over the obligate aerobic filamentous microorganisms in the aeration tank. However when biocenoses composition in the following examples is taken into consideration it is seen that concentration of the filamentous microorganisms is appreciably higher in completely mixed activated sludge.

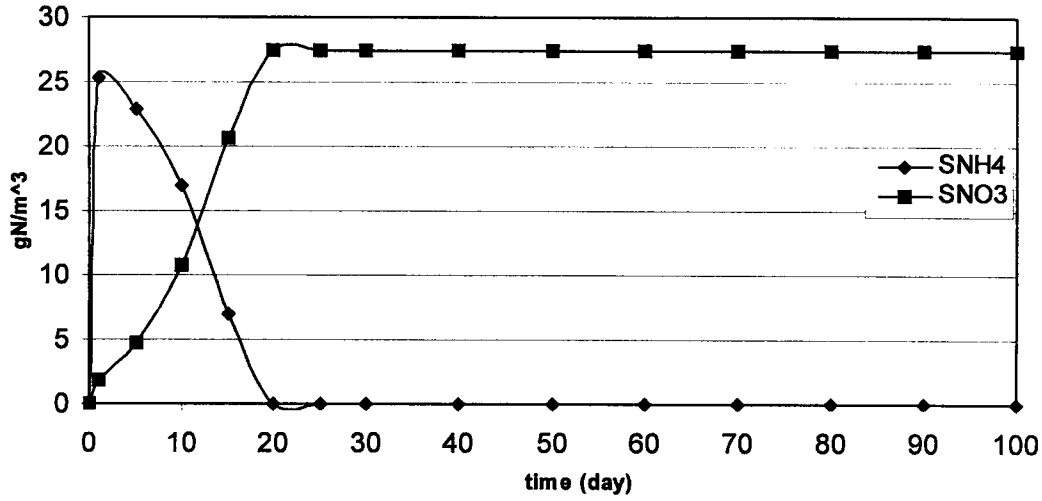

Figure 4.11 S_{NH4} and S_{NO3} composition in the completely mixed aerobic reactor as predicted by AEROFIL in AQUASIM

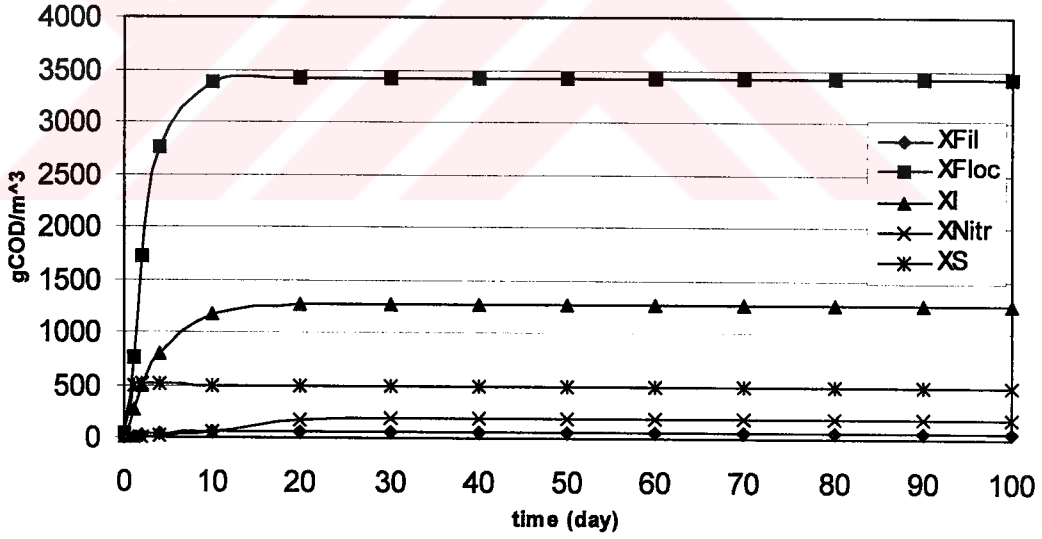

Figure 4.12 X_{Fil}, X_{Floc}, X_I, X_{Nitr}, X_S composition in completely mixed aerobic reactor as predicted by AEROFIL in AQUASIM

4.12 Predicting Population Distribution in A/O (Anaerobic/Oxic) Process

Simulation of floc biocenoses and substrate removal in A/O process as shown in Figure 2.2 was carried out by AEROFIL model in AQUASIM program according to the data in example 8, in Table 4.1. A short SRT of 3 days was chosen to exclude nitrifiers and nitrate in the recirculation. Thus nitrate in the anaerobic tank was avoided. Oxygen was also avoided in this tank by omitting aeration.

The carbonaceous matter composition in the aerobic tank is depicted in Fig. 4.13, as predicted by AEROFIL with AQUASIM dynamically. However dynamic output eventually reaches steady-state as indicated by the steady output values. From this figure it can be seen that soluble inert COD remains constant for most of the time. Initial increase in S_H, which is conceived as included in X_S in ASM1, is evidently increasing initially due to the start-up process where S_H was produced progressively as biomass increased and hydrolyzed into soluble degradable matter. Upon completion of the start-up S_H, and readily biodegradable substrate from the influent, S_S, approaches to zero as degradation proceeds. The reason of the decrease of the S_H and S_S is the aerobic growth of microorganisms.

The Figure 4.14 depicts the ammonium-nitrogen S_{NH4} and nitrate-nitrogen S_{NO3} concentrations in the aerobic tank. As there is no nitrification taking place in the aerobic tank, S_{NH4} concentration is high and S_{NO3} concentration remains low in the system. From these figures it can be seen that C removal is achieved while N removal did not take place according to the process objectives.

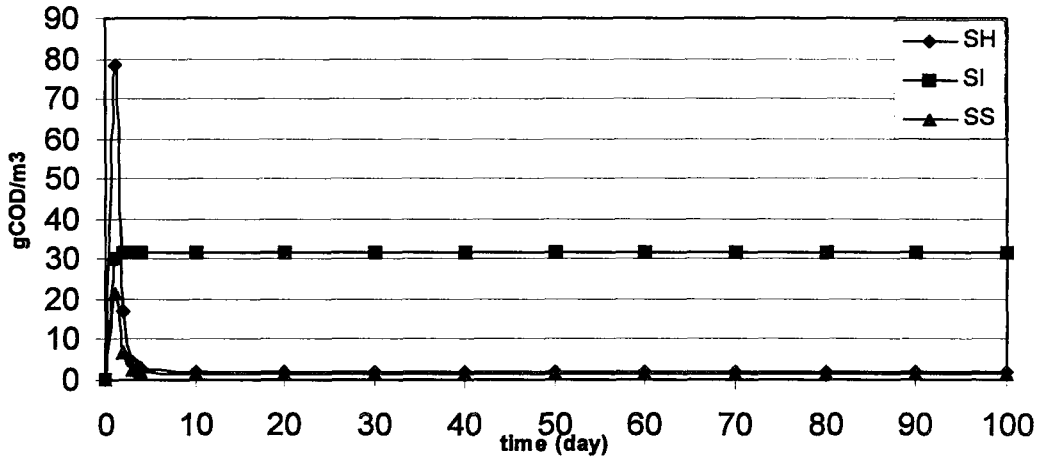

Figure 4.13 The S_H, S_I and S_S concentrations in the aerobic tank of A/O Process as predicted by AEROFIL in AQUASIM

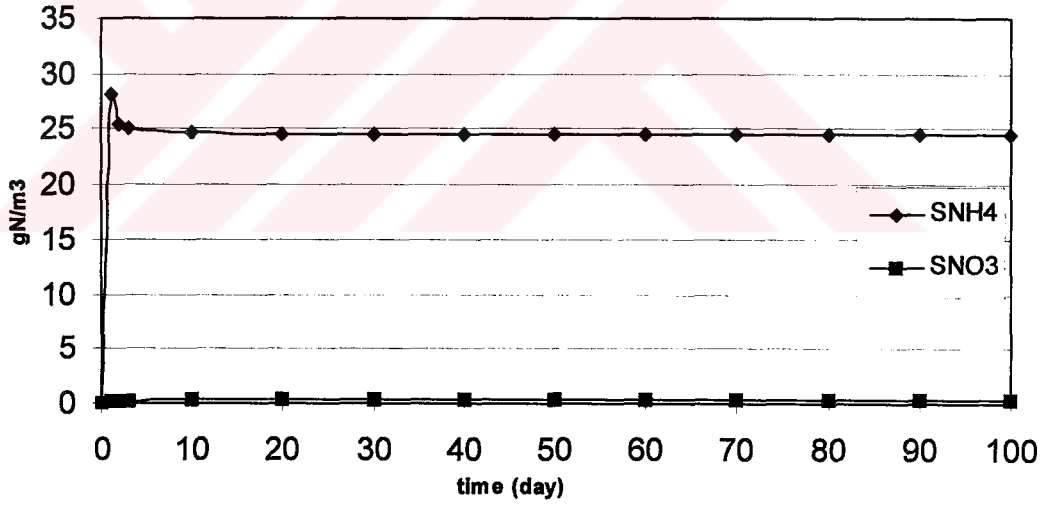

Figure 4.14 S_{NH4} and S_{NO3} composition in the aerobic tank of A/O Process as predicted by AEROFIL in AQUASIM

The Figure 4.15 shows the biocenoses composition in the aerobic tank. $X_{Fil}/X_{Biomass}$ (%) ratio was calculated as 1.298. As the first tank is anaerobic, growth of the obligate aerobic filamentous microorganisms in the system is effectively suppressed.

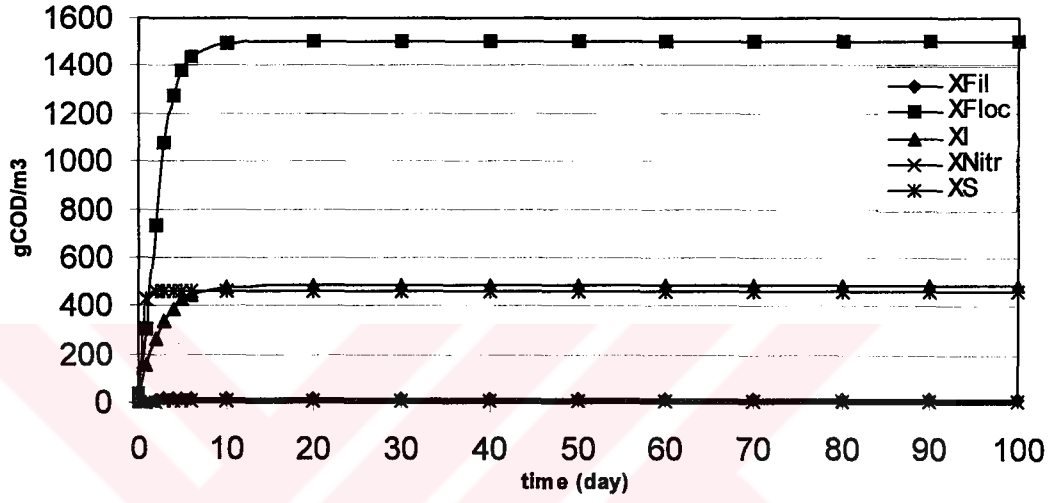

Figure 4.15 X_{Fil}, X_{Floc}, X_I, X_{Nitr}, X_S composition in the aerobic tank of A/O Process as predicted by AEROFIL in AQUASIM

4.13 Predicting Population Distribution in A^2/O (Anaerobic/Anoxic/Oxic) Process

The dynamic simulation of A^2/O process shown in Figure 2.3 for substrate removal and biocenoses composition was carried out by AEROFIL model and using AQUASIM according to plant characteristics and wastewater composition given in

example 9 in Table 4.1. During design of three-stage systems such as A^2/O process, overall size for C and N removal is determined by considering about 12% anaerobic zone, 38% anoxic zone and 50% aerobic zone. Nitrate recirculation to the anoxic tank is equal to Q_{in}.

In Figure 4.16, it can be seen that soluble inert COD, S_I, remains constant while hydrolysis products, S_H, and readily biodegradable substrate, S_S, approaches to zero due to aerobic growth of microorganisms. The ammonium-nitrogen, S_{NH4}, and nitrate-nitrogen, S_{NO3}, concentrations in the aerobic tank are shown in Figure 4.17. As expected S_{NH4} concentration decreases and approaches to zero while S_{NO3} concentration increases. Effluent contains less than 10 mg/l nitrogen in this case. The biocenoses composition in the aerobic tank is shown in Fig. 4.18. $X_{Fil}/X_{Biomass}$ (%) ratio was calculated as 0.2054. As the first tank is anaerobic and the second one is anoxic, growth of the obligate aerobic filamentous microorganisms is largely suppressed in this system.

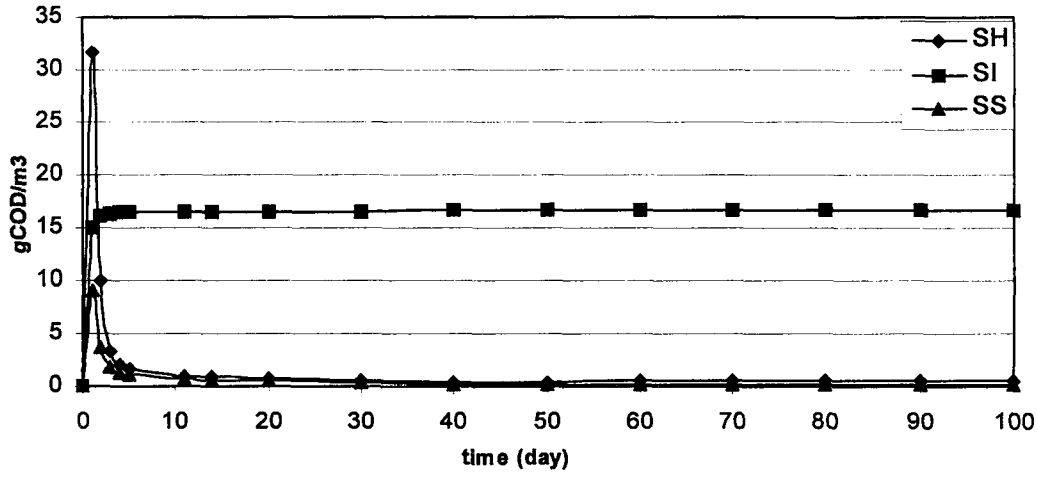

Figure 4.16 S_H, S_I and S_S composition in the aerobic tank of A^2/O Process as predicted by AEROFIL in AQUASIM

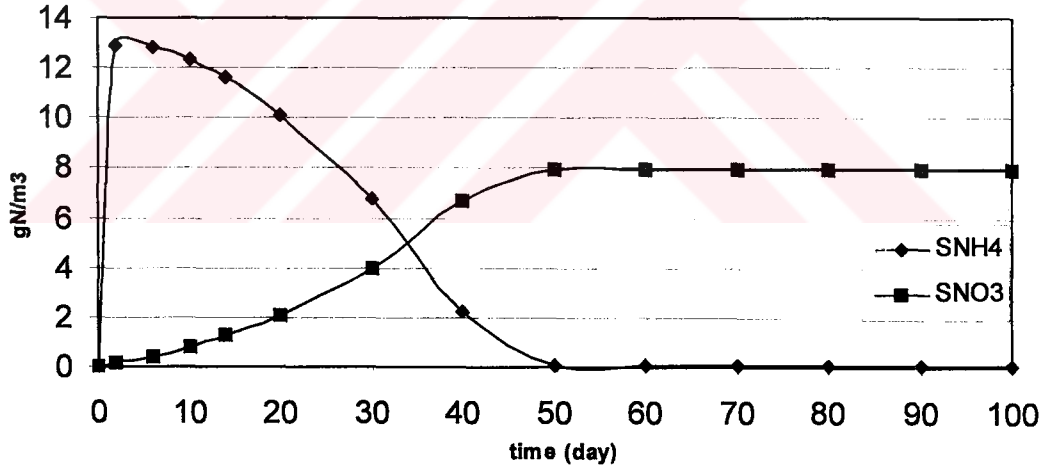

Figure 4.17 S_{NH4} and S_{NO3} composition in the aerobic tank of A^2/O Process as predicted by AEROFIL in AQUASIM

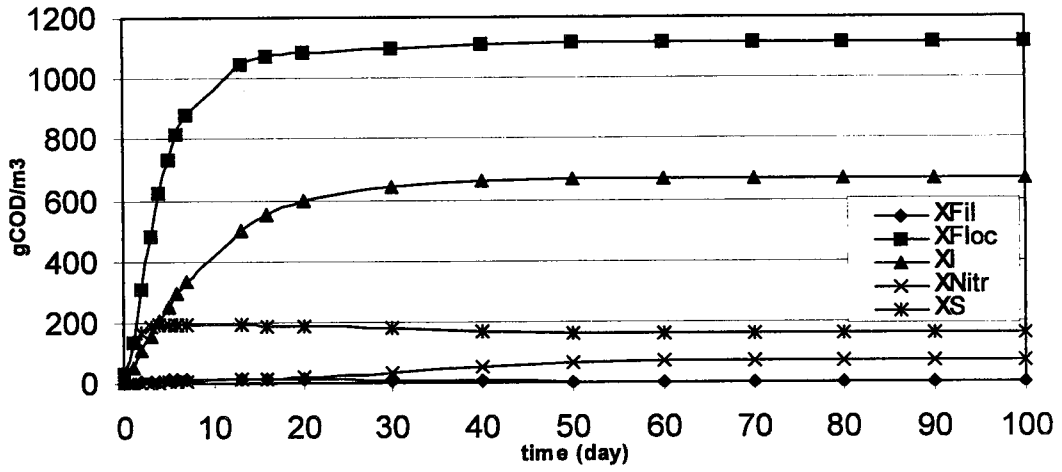

Figure 4.18 X_{Fil}, X_{Floc}, X_I, X_{Nitr}, X_S composition in the aerobic tank of A^2/O Process as predicted by AEROFIL in AQUASIM

4.14 Predicting Population Distribution in Bardenpho Process

The dynamic simulation of Bardenpho process shown in Figure 2.3 for substrate removal and biocenoses composition predictions were carried out by AEROFIL model and using AQUASIM according to plant characteristics and wastewater composition given in example 10 in Table 4.1. Anoxic tank size was taken as 20% of the total tank volume and remaining volume, 80%, was aerobic. Nitrate re-circulation flow rate was equal to Q_{in}.

The carbonaceous matter concentration in the second aerobic tank is depicted in Fig. 4.19. From this figure it can be seen that the concentration of soluble inert COD, S_I, remains constant most of the time. The initial rise of S_I was actually due to start-up

of the reactor. The readily biodegradable substrate, S_S, approaches to zero due to rapid uptake but hydrolysis products, S_H, increases to some extent and remains constant after 30 days upon when steady-state conditions truly establishes in the plant. Somewhat higher concentrations of the hydrolysis products, S_H, may be due to reduced anoxic hydrolysis rate under anoxic conditions. This is reflected by the η_h value employed in the model. Same should also be true for the other types of processes

Ammonia-nitrogen S_{NH4} and nitrate-nitrogen S_{NO3} concentrations in the second aerobic tank are shown in Figure 4.20. As the wastewater leaves the anoxic stage and enters the final aerobic tank, the residual S_{NH4} gets oxidized to nitrate. Hence S_{NH4} approaches to zero and S_{NO3} increases to maximum. It is evident from this figure that more re-circulation or additional carbon source is required in the system for higher N-removals.

The biocenoses composition in the second aerobic tank is shown in Fig. 4.21. $X_{Fil}/X_{Biomass}$ (%) ratio was calculated as 0.3838. As there is a pre-anoxic stage prior to the aerobic stage, growth of obligate aerobic filaments is largely discouraged in this system too.

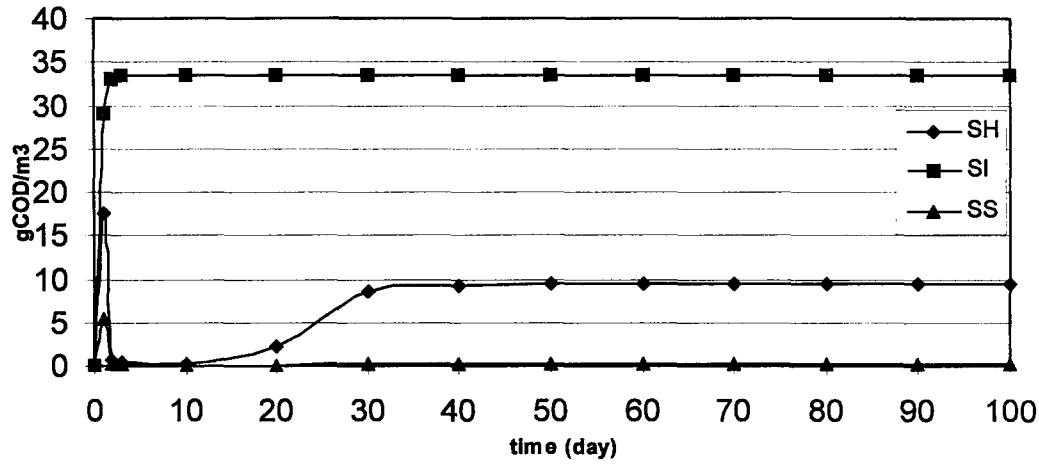

Figure 4.19 S_H, S_I and S_S composition in the 2nd aerobic tank of Bardenpho Process as predicted by AEROFIL in AQUASIM

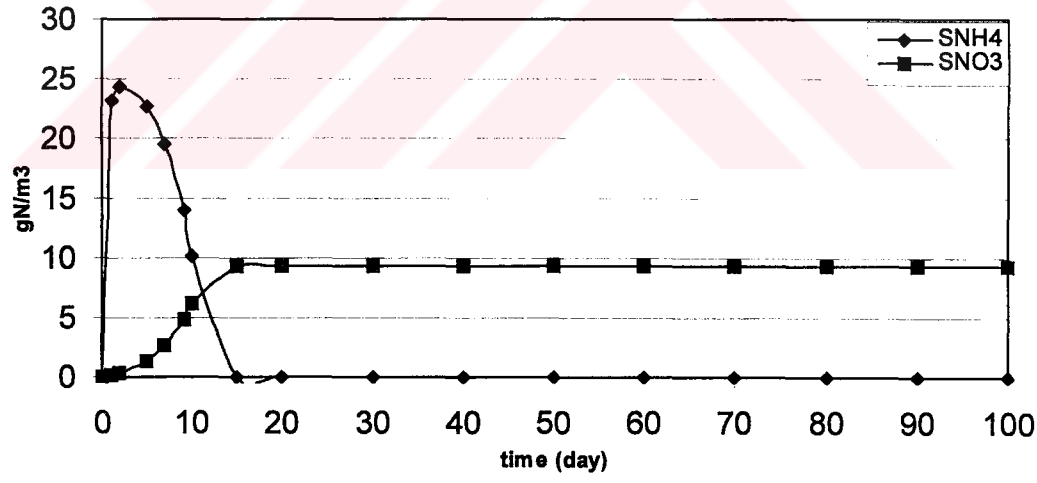

Figure 4.20 S_{NH4} and S_{NO3} composition in the 2nd aerobic tank of Bardenpho Process as predicted by AEROFIL in AQUASIM

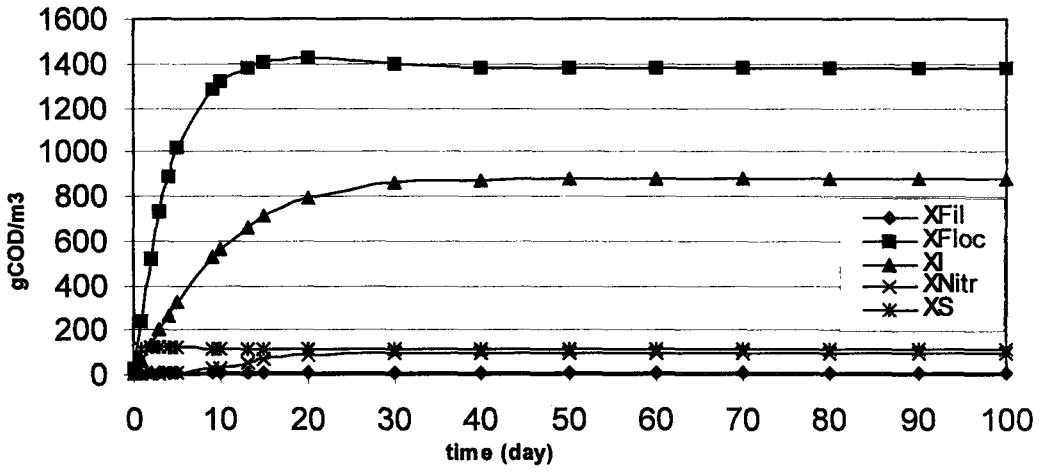

Figure 4.21 X_{Fil}, X_{Floc}, X_I, X_{Nitr}, X_S composition in the 2nd aerobic tank of Bardenpho Process as predicted by AEROFIL in AQUASIM

CHAPTER 5

CONCLUSION AND RECOMMENDATIONS

In this thesis activated sludge population dynamics was studied by using AEROFIL model coded in AQUASIM utility program. In the first part of the study, AEROFIL model components were coded into the AQUASIM program, and was then run for simulation. In the second part of the study, model was verified and supported for different plant characteristics and activated sludge modifications by using the literature data. Seven different sets of plant characteristics in the literature, including influent flow rate, influent substrate concentrations, solids retention times, temperature and reactor volume were used as model supporting data one of which is used for model verification. The following conclusions were drawn as a result of these studies:

5.1 Conclusion

1. Three forms of microorganisms were identified for modeling of activated sludge population dynamics. These being facultative aerobic floc-formers, X_{Floc}, obligate aerobic filamentous microorganisms, X_{Fil}, and nitrifying microorganisms, X_{Nitr}.

2. Unlike in ASM1, readily biodegradable substrate in the influent, S_S, and hydrolysis products, S_H, were distinguished. This is necessary for the description of competition between microorganisms. Besides, particulate slowly biodegradable substrate, X_S, and influent microorganisms concentration in the wastewater must be defined.

3. Floc-formers represent a significant part of the biocenoses. Hence, their kinetic parameters can be determined with batch test directly. But, filamentous microorganisms represent only a small part of the biocenoses. Estimation of their kinetic parameters by lab experiments is exceedingly difficult, if at all possible, hence may only be determined indirectly. Kinetic parameters of nitrifiers can also be determined experimentally as the floc-formers.

4. Readily biodegradable substrate from the influent favors growth of filamentous microorganisms whereas readily biodegradable hydrolysis products favor growth of floc-formers due to different diffusional resistances.

5. Introduction of high concentrations of readily biodegradable substrate into the completely mixed activated sludge causes increase in the sludge volume index, which results in sludge bulking. For example an increase in readily biodegradable substrate from 10 $gCOD/m^3$ to 60 $gCOD/m^3$ caused an increase in sludge volume index from 50 ml/g to 400 ml/g.

6. Changing influent pattern from step feed to head-end-feed (plug flow) improved settling characteristics. For example, in the head-end-feed, which was simulated by cascades of completely mixed reactors, filamentous microorganism to total biomass ratio, % $X_{Fil}/X_{Biomass}$, decreased from 3 to 1 after four cascade reactors. Further number of cascades decreased the ratio up to 0.5.

7. Introduction of floc-former microorganisms into a completely mixed activated sludge caused reduction in % $X_{Fil}/X_{Biomass}$ ratio up to 0.5.

8. Introduction of filamentous microorganisms into the completely mixed activated sludge in the feed caused % $X_{Fil}/X_{Biomass}$ ratio to increase up to 15. The ratio was zero when feed filamentous microorganism was 0 gCOD/m^3. The RBCOD in the feed caused remarkable increase in the ratio.

9. The % $X_{Fil}/X_{Biomass}$ ratio decreased to 0.8 in the completely mixed activated sludge upon inclusion of 10% relative aerobic selector volume (V_{sel}/V_{tot} (%)); as compared to 5 at equivalent conditions without selector.

10. Compartmentalization of aerobic selectors also caused considerable reduction in % $X_{Fil}/X_{Biomass}$ ratio; as the ratio decreased to 0.38 compared to 0.66 obtained with single selector compartment.

11. With the introduction of anoxic selectors, % $X_{Fil}/X_{Biomass}$ ratio decreased to 0.3 at 10% relative selector volume as compared to 5.2 obtained in no selector condition.

12. Compartmentalization of anoxic selectors was not effective since obligate aerobic filamentous microorganisms were unable to grow under anoxic conditions.

13. Floc biocenoses was also modeled for some of the advanced nutrient removing activated sludge processes such as A/O, A^2/O and Bardenpho Processes. A completely mixed activated sludge process was also modeled under identical conditions to serve as control. The computed % $X_{Fil}/X_{Biomass}$ ratios were 1.298, 0.2054, 0.3838 and 1.501 respectively. The ratio was lowest in A^2/O Process since obligate aerobic filamentous microorganisms were subjected consecutively to anaerobic and anoxic conditions prior to aerobic conditions. Ratio was highest in the completely mixed activated sludge system.

14. The AEROFIL model may successfully predict the activated sludge biocenoses and the AQUASIM program is a useful tool in simulating activated sludge systems.

5.2 Recommendations

It is now clear that the AEROFIL model used via AQUASIM program can simulate activated sludge population dynamics for different plant configurations. However experimental data is needed on biokinetic constants and wastewater characteristics for experimental verification.

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