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ADVANCED WASTEWATER TREATMENT

Report No 4.

FACTORS INFLUENCING SLUDGE SETTLING PARAMETERS AND SOLIDS FLUX IN THE ACTIVATED SLUDGE PROCESS
A Literature Review

Anna Stypka

Stockholm 1998

Joint Polish - Swedish Reports

E. P³aza, E. Levlin, B. Hultman
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Preface

This report series on "Advanced wastewater treatment" has been realized as a part of the collaboration between the Technical University of Cracow and the Royal Institute of Technology, KTH, in Stockholm. Three earlier reports are:

Report No I. Phosphorus removal from wastewater. A literature review.
Stanislaw Rybicki, 1997.
Report No 2. Proceedings of a Polish-Swedish seminar, KTH, May 30, 1997.
Report No 3. Proceedings of a Polish-Swedish seminar, Nowy Targ,
October 1-2, 1998.

As a part of this cooperation Mrs. Anna Stypka has had a possibility to stay 1.5 months in summer 97 and 4 months in fall 1998 as a guest researcher at the Royal Institute of Technology. During her visits he studied factors influencing sludge settling parameters and solids flux in the activated sludge process and her work resulted in this literature review presented in this report No 4. The work will be partly utilized in a future doctorate thesis that is being conducted by Mrs. Stypka at the Cracow University of Technology.

Financial support has been obtained from the Swedish Institute (SI). Bengt Hultman has provided advises in outlining the project, helped in the collection of information and acted as supervisor. Elzbieta Plaza and Jozef Trela provided valuable comments and support throughout the project.

Stockholm, December 1998

Bengt Hultman

Associate professor

Abstract

This report was aimed to provide a summary of significant developments that have been taking place in the area of defining activated sludge settling characteristics and the influence it has on the design and operation of secondary clarifiers. The report presents an overview of process disturbances encountered during the operation of the activated sludge process focusing mostly on those resulting from sludge settling properties (floc, bulking sludge, rising sludge, foaming). Special attention was given to the sludge biomass characteristics reflecting its microbiological structure, filamentous content and bioflocculation ability. Defining the conditions for biological selection of floc orming microorganisms in activated sludge systems with and without nutrients removal have also been discussed.

Additionally, chemical and physical properties of wastewater flocs such as: strength, density, settleability, filterability and water binding have been presented together with different methods of their evaluation. A relationship between such properties and sludge settleability has been discussed.

Secondary clarifiers play an important role in the performance of the activated sludge process. They combine functions of: a thickener , clarifier and a storage tank to store the sludge during the peak flows. Therefore, many different factors influence their performance and secondary clarifiers should in design take into consideration all these functions. Accurate prediction of the secondary clarifier performance requires establishing sound measures of solids settleability. The report discusses the advantages and shortcomings of different sludge indices (SVI, SSVI, SQI, DSVI) and their relationships with zone settling velocity. Also a brief overview of development of the flux theory has been presented in the report since this approach has often been used in the design and operation of secondary clarifiers. Different mathematical models have been reviewed of the secondary clarifies performance both as a separate unit and also combined to represent the entire activated sludge system.

TABLE OF CONTENTS

PREFACE i

ABSTRACT iii

1. INTRODUCTION 1

2. GENERAL DESCRIPTION OF THE ACTIVATED SLUDGE PROCESS 2

2.1. Historical aspects 2

2.2. Removal of organic material 3

2.3. Nitrification 4

2.4. Denitrification 6

2.5. Biological phosphorus removal 6

2.6. Clarification function in the sedimentation basin 7

2.7. Thickening function in the sedimentation basin 11

3. PROCESS DISTURBANCES IN THE ACTIVATED SLUDGE PROCESS 16

3.1. Sludge activity 17

3.2. Hydraulic disturbances 18

3.3. Sludge properties 19
3.3.1. Sludge bulking 20
3.3.2. Dispersed sludge (pin-point sludge) 34
3.3.3. Foaming 35

4. SLUDGE CHARACTERIZATION 42

4.1. Biochemical characterization 42
4.1.1. Exocellular polymers (ECP) 42

4.2. Physical chracteristics of activated sludge flocs 48
4.2.1. Floc size. 49
4.2.2. Density 53
4.2.3. Floc strength 55
4.2.4. Surface charge 57

4.3. Microscopical methods 58

4.4. Sludge Indices **61**
4.4.1. Sludge Volume Index (SVI) 61
4.4.2. Stirred Specific Volume Index (SSVI) 63
4.4.3. Diluted Sludge Volume Index (DSVI) 64
4.4.5. Sludge Quality Index (SQI) 65
4.4.6. Modified Sludge Index (I_{VF}) 66
4.4.7. Bulking Index (BI) 66
4.4.8. Relationships between different sludge indices 67

4.5. Sedimentation curves. **69**

5. MODELING OF CLARIFICATION AND THICKENING FUNCTIONS **74**

5.1. General background **74**
5.1.1. Batch settling 74
5.1.2. Continuous settling 77
5.1.3. Thickening of activated sludge 78

5.2. Solids flux theory **81**

5.3. Settling velocity models **82**

5.4. Influence of sludge indices on solids flux **84**
5.4.1. Foster Model 85
5.4.2. Pitman, Ekama and Marais Model 85
5.4.3. Daigger and Roper Model 86
5.4.4. Wahlberg and Keinath Model 87
5.4.5. Cho Model 88
5.4.6. Daigger Model 89

5.5. Clarification and Thickening Models **90**
5.5.1. Settling and thickening zone 90
5.5.2. Compression zone height 98

CONCLUSIONS **100**

REFERENCES **103**

FACTORS INFLUENCING SLUDGE SETTLING PARAMETERS AND SOLIDS FLUX IN THE ACTIVATED SLUDGE PROCESS

A Literature Review

1. Introduction

In the activated sludge process, the quality of the effluent is to a large extent dependent on how the biological sludge mass can be separated from the treated wastewater. The solids/ liquid separation traditionally takes place through a separation of spontaneously aggregated flocs of activated sludge by gravity sedimentation in secondary clarifiers .

The secondary clarifiers combine the functions of: (1) a thickener - to thicken the returned sludge to the required concentration, (2) a clarifier - to produce a clear effluent and (3) a storage tank to store sludge during peak flows. All these three functions should be performed at the level required by the design, or otherwise suspended solids would escape from the clarifier to the effluent and increase the final BOD concentration. The behavior of the secondary clarifiers is influenced mostly by their design features (surface area, depth, inlet and outlet configuration, hydraulic regimes and disturbances) and sludge characteristics (its settleability and thickenability) so both these factors should be carefully examined while reviewing the secondary clarifier performance.

Specially activated sludge characteristics, affecting its ability to flocculate, settle and thicken seems to be very complex and depend on many variables. The various factors like: amount and type of filamentous microorganisms, floc size, specific surface area, surface charge, amount of extracellular polymers, amount of divalent cations (e.g. Ca^{2+}, Mg $^{2+}$), floc strength, floc density and hydrophobicity are responsible for an efficient settling process.

In recent years several advances have been made in understanding bulking phenomena and the importance of the environmental factors on sludge settling characteristics. Settling characteristics are thought to be strongly influenced by reactor conditions (low nutrient and oxygen levels, feed pattern) through biomass population dynamics and surface chemistry. An imbalance between various filamentous and floc forming organisms has been correlated to activated sludge settleability. Much research has been done to characterize the desired balance and to understand the factors that encourage one or more of these organisms to grow out of balance with the others. Specially right now when many wastewater treatment plants are being expanded to accommodate the nutrients removal processes the activated sludge process is often carried out at higher solids concentrations and with alternating anaerobic, anoxic and aerobic conditions. All these reasons stress the importance of an extensive research towards better understanding the factors controlling sludge bulking.

This literature review is a summary of the factors, both physical and biological, known to affect the formation of activated sludge flocs and the principles behind the sedimentation process. Some microbiological and kinetics aspects related to bacterial selection have been reviewed in this report as well as the main mechanisms involved in the competition of filamentous and floc-forming bacteria. The hydraulic regimes and design criteria are treated in brief while most attention is focused on sludge settling characteristics and solids flux theory review and their application for design and operation of secondary clarifiers.

2. General description of the activated sludge process

2.1. Historical aspects

The activated sludge process has already been used in practice for almost eighty years. Originally, it was developed in England by Arden and Lockett in 1914 (Metcalf and Eddy, 1979) and since then it has been subjected to many improvements throughout the years. It is rather a unique biotechnological process which consists of an aerated suspension of mixed bacterial cultures which carry out the biological conversion of the contaminants in the wastewater. The aeration tank, while having many possible configurations, basically retains the influent wastewater for a number of hours (or days) in a well mixed/aerated environment, before forwarding the effluent for further settling to the secondary clarifier. The end products of the clarification process are clarified effluent that is discharged to the open water bodies and sludge.

A fraction of the sludge is returned to the aeration tank and is called returned activated sludge. The sludge contains a high density of biomass and an active population of microorganisms is always maintained in the tank. The influent wastewater provides the basic food source for the microorganisms in the aeration tank. This biodegradable organic material is converted into new bacterial cells and other end products include CO_2, NO_3 and SO_4.

Typical features of the activated sludge include (Eckenfelder and Grau, 1992):

- Diverse substrate in terms of chemical composition and variety of particle sizes;
- Multispecies biological culture, desirably growing in aggregates (flocs);
- Widely fluctuating flows, temperatures, and changes in the influent wastewater concentration and composition;
- Ability to metabolize a vast number of organic compounds and to oxidize/reduce/ polymerize etc. compounds containing nitrogen, phosphorus, sulfur and others;
- A variety of reactor configurations used e.g. completely stirred tanks, plug-flow, sequencing batch reactors, oxic, anoxic, and anaerobic selectors;

The activated sludge process exists in a large number of modifications and variations. Over last years the most important development in practice can be observed in industrial wastewater treatment, nutrient removal (N and P) and bulking control technologies.

2.2. Removal of organic material

When a biodegradable organic food source is supplied to a heterotrophic (utilizing organic matter for energy, as distinct from autotrophic which use CO_2 as the carbon source) microorganism population in a well-aerated environment the following phenomena occur (adapted from Kiely, 1997):

- The readily soluble biodegradable particulate COD goes through the cell wall and is metabolized quickly;
- The slowly biodegradable particulate COD is adsorbed on to the organisms and stored. This action removes all the particulate and colloidal COD which than over time is broken down and transferred through the cell wall and metabolized;
- Some of the COD metabolized is converted to new cells while the remainder is lost as heat in the energy process required for the new cell synthesis;

- At the same time there is a net loss of biomass, termed endogenous mass loss, where some of the organisms utilize as food their own stored food materials and dead cells.

Growing cells utilize external substrate and external additional nutrients as required for growth and energy. The biochemical equation for bacterial cell respiration and synthesis in using organic matter as substrate in the activated sludge process is:

$$\text{org.matter} + O_2 + \text{nutrients} \xrightarrow{\text{bacteria}} CO_2 + NH_3 + \text{new biomass} + \text{end products}$$

Organisms have developed a number of biomechanisms enabling them to survive rather long periods without external substrate. Several pathways and biomechanisms enabling them to survive have been recognized; some more may be still unknown (Eckenfelder and Grau, 1992). First, bacteria can store products rich in both energy and nutrients. Second, some of the cell materials can serve several needs. They can be either utilized for cell growth, if the nutritional and environmental conditions are favorable, or they can be utilized as internal sources of energy (endogenous substrate) during starvation periods. These phenomenas are of an increased importance to wastewater treatment since they are employed in several successful modern technologies.

The activated sludge process is governed by the microorganism characteristics and the physical configuration of the aeration tank. As such, the biological kinetics and the process kinetics are closely interconnected. Process kinetics is based on the Monod equation and it has been described by many authors (Metcalf and Eddy, 1979; Eckenfelder and Grau, 1992).

2.3. Nitrification

The main objective of wastewater treatment is stabilization of the carbonaceous matter. However, the nutrients as nitrogen and phosphorus contribute greatly to eutrophication of the receiving wasters and many countries have passed legislation to remove those compounds from wastewater. Nitrogen in wastewater is generally in the forms of organic N and ammonia N, both in soluble and particulate forms; wastewater contains normally insignificant amounts of nitrite and nitrate N. Organic N and ammonia N are undesirable in wastewater effluents since they both exert a nitrogenous oxygen demand and ammonium N is also toxic to fish life.

Nitrification is the biological process where ammonia serves as a substrate for nitrifying bacteria that oxidize ammonium to nitrite and nitrate in two steps as follows:

$$NH_4^+ + 1.5\ O_2 \longrightarrow NO_2^- + H_2O + 2H^+ + 250\ kJ \qquad (1)$$

$$NO_2^- + 0.5O_2 \longrightarrow NO_3^- + 75\ kJ \qquad (2)$$

All acidity and most energy are produced in the first step, called nitritation. The second stage is called nitratation. Theoretical oxygen demand of nitrification is determined according to the above equations :

$$TOD_{nit} = 4.57\ g\ O_2\ /g\ NH_4 \qquad (3)$$

Those equations do not consider the synthesis of biomass and the utilization of CO_2 by the nitrifiers. If those phenomena are considered, TOD_{nit} is between 4.25 and 4.57 g/g. Some nitrifiers as *Nitrosomonas, Nitrococcus* are responsible for the first step while *Nitrobacter* is a typical representative of the second step. Nitrifiers utilize CO_2 as a source of carbon. Typically their growth rate can be described by the following equation (Eckenfelder and Grau, 1992):

$$\mu = \mu_{max} \frac{S_{NH}}{S_{NH} + K_{NH}} \frac{S_{CO_2}}{S_{CO_2} + K_{CO_2}} \frac{S_{O_2}}{S_{O_2} + K_{O_2}} \qquad (4)$$

where: S_{NH}- concentration of NH_4^+

S_{CO2} - concentration of CO_2

S_{O2} - concentration of dissolved oxygen

K_{ii} - appropriate saturation coefficients

Nitrification efficiency depends highly on the dissolved oxygen concentration and the wastewater alkalinity. Because during the first reaction step the alkalinity is being used up, there may be not enough buffering capacity within the system. In order to maintain sufficient nitrification rate dissolved oxygen concentration has to be maintained at 1.5 to 2 mg/l and alkalinity at the level of at least 1 - 1.5 mmol/l (Eckenfelder and Grau, 1992).

2.4. Denitrification

In anoxic environment many bacteria can reduce nitrate to nitrogen gas (N_2) when electron donors are present. This process is referred to as denitrification. The most common denitrification reactions are accomplished by heterotrophic bacteria (*Pseudomonas, Micrococcus, Bacillus* and *Alcaligenes*), which rely on organic compounds such as i.e. methanol as electron donors:

$$NO_3^- + 0.83\ CH_3OH\ = 0.5\ N_2 + 0.83CO_2 + 1.17H_2O + OH^- \qquad (5)$$

The nitrogen gas produced is released to the atmosphere; therefore it is removed from the wastewater.

Heterotrophic denitrification is often accomplished in wastewater treatment by including anoxic zones in the activated sludge process. Single, double and triple sludge systems have been employed. However, the single sludge system is often considered to be the best activated sludge modification for biological nitrogen removal. In addition it does not require the external carbon source that is necessary for double and triple sludge systems. A single sludge nitrogen removal system is comprised of an initial anoxic stage, which facilitates nitrate reduction to nitrogen gas and BOD removal, followed by an aerobic tank, for continuous removal of BOD and oxidation of ammonia to nitrate. A portion of the mixed liquors is recycled to the anoxic tank, which provides nitrate to maintain anoxic conditions.

2.5. Biological phosphorus removal

According to the literature review conducted by Rybicki (1997) the evolution of biological phosphorus removal started with Levin and Shapiro (1965) who reported on extensive investigations into phosphorus uptake and release. They named the observed high phosphorus removal " a luxury uptake". They also observed that uptake and release of phosphorus are reversible processes; an observation of great importance for the development of the process.

After many years of an extensive research it was found that under certain anaerobic - anoxic conditions, activated sludge microorganisms can contain more then the normal 2-3% phosphorus. In fact, some forms of bacteria called bio-P bacteria, most notably *Acinetobacter* species, can store as much as 5-6 % of VSS as phosphorus.

According to the available biological models (see Rybicki, 1997), simple carbonaceus substrates are transported across the cell wall of bio-P bacteria in the anaerobic reactions, and stored as poly-ß-hydroxybutyrate (PHB) that serves as an energy reserve. Energy for this process is obtained from hydrolysis of poly-P chains accumulated in the organisms. Under aerobic conditions energy produced from the consumption of carbon (stored or external) will result in the growth and accumulation of phosphate by bio-P bacteria. Bio-P systems operate most effectively when a major portion of carbonaceous substrate is in form of volatile fatty acids such as acetic and propionic acid.

Due to such a metabolic pathway bacteria with large reserves of phosphate, such as bio-P bacteria have a distinct advantage when anaerobic storage of substrate is necessary. Therefore, a preferential selection of bio-P bacteria under suitable sequence of anaerobic-aerobic conditions occurs. A typical bio-P system is composed of an anaerobic stage (all nitrate and dissolved oxygen excluded) in which substrate is sorbed by bacteria and phosphate are released. This is followed by an aerobic stage in which stored PHB is oxidized and large quantities of phosphate are removed from wastewater.

2.6. Clarification function in the sedimentation basin

In the optimal design of an activated sludge system there is a relationship between the design of the aeration tank and the secondary clarifier. The performance of the aeration tank varies with the return sludge concentration and flow rate while the clarification and thickening functions of the clarifier depend on the effluent from the activated sludge tank. Design of the secondary clarifier is generally based on either its function of clarification or thickening. Clarification efficiency of a secondary clarifier is a critical factor in determining the efficiency of the entire wastewater treatment system. However, the clarification criterion is important only at severe changes in hydraulic load and then only for a short period of time, because if the hydraulic loading persists the thickening or solids handling criterion will in time become the governing one (Laquidara and Keinath, 1983).

Traditional designs of secondary sedimentation basin for clarification function used surface loading as design parameter. Hazen (1904) developed an ideal horizontal flow model to describe the clarification process. It is valid for discrete particles. In principle it means that all particles with the settling velocity larger than v_f will settle while passing an ideal settler from one end to the other (assuming that all particles move with identical velocity vectors).

$$v_f = Q/A \tag{6}$$

where: v_f = settling velocity, m/s; Q = flow, m^3 /s; A = sedimentation area, m^2.

The ratio Q/A is called the overflow rate (or surface loading) and it sets the capacity limit for the process (for an idealized settler). For a continuous settler, the depth of the settler and the detention time should be such that:

$$v_f = \text{depth/detention time} \tag{7}$$

When the capacity limit is exceeded, only a fraction of the flocs will be separated; i.e. the particles in the lower region of the influent stream (assuming that particles of various sizes are uniformly distributed over the entire depth of the basin at the inlet). This theory requires that the flow is uniformly distributed over the whole cross-section area of the sedimentation tank.

Camp (1953) made a modification of Hazen´s theory to take into account settling velocity distribution. He considered the trajectory of a discrete non-flocculent particle in an ideal sedimentation tank. His analysis demonstrated that the surface overflow rate was equal to the sedimentation velocity of the slowest settling particle to be completely removed. Given the fact that the solids concentration in the upper layers of the clarifier is sufficiently low to result in discrete particles, it can be assumed that this theory can be extended to the clarification component of primary and secondary clarifiers. However, the problem lies not in the settling velocity theory but in the flocculent nature of these particles which affects the particle size distribution from one layer to another.

Since then researchers tried to develop models that would incorporate basin turbulent transport coefficients, particle fall velocity and basin resuspension. One of the models was the model developed by Larock et al. (1983) where the resulting distribution of particle concentration throughout the basin and the basin particle removal efficiences were computed as a function of particle fall velocity and the amount of particle deposition on the basin floor. Li and Ganczarczyk (1987) have found the settling velocity of activated sludge floc particulates to be a linear function of their cross-sectional diameter. However, little effort has been made to describe the size distribution of the solids in the upper layers of a settling tank as a function of the operational characteristics of the activated sludge process.

In contrast Fitch (1957) maintained that hydraulic detention time rather than overflow rate was the major variable governing solids removal from secondary clarifiers. His conclusions was based on experiments with a suspension of calcium carbonate pumped at different rates to settling columns of different heights. A plot of detention time versus overflow rate indicated that percent removal was governed more by detention time than by overflow rate. In his later works Fitch (1979) proposed that for a flocculent suspension clarification is a two-step process: first the particles flocculate and then the flocculates settle out. If the flocculation is rapid (or the suspension is pre-flocculated before entering the settler), i.e. the settling velocity is constant over the entire sedimentation time, the surface area of the settler will govern the removal of solids. If the flocculation is slow, the removal will be governed by the flocculation kinetics and thus the retention time (and thereby dependent on the tank depth).

Pflanz (1969) carried out a 2- year study of secondary sedimentation on one circular and two rectangular full- size tanks. Overflow rate, the concentration of solids in the feed to the settler, sludge settleability as measured by the sludge volume index (SVI) and temperature were all found to affect the effluent clarity. Because of the concentration of suspended solids in the mixed liquor was the most important factor, Pflanz recommended that the secondary settlers be designed on the basis of mixed liquor suspended solids (MLSS) concentration as well as on hydraulic loading.

Examples of design criteria based on empirical hydraulic or solids loading rates are those recommended by: Institute for Water Pollution Control (Table 1), United States Environmental Protection Agency (Table 2) WEF-ASCE design manual (Table 3).

Table 1. Secondary settling tank design criteria for activated sludge plants set by the IWPC (1973)

1. Maximum overflow rate at peak wet weather flow	1.0 m/h
2. Minimum retention time at peak dry weather flow	1.5 h
3. Maximum weir loading rate at peak dry weather flow	8.3 $m^3/h\ m^2$

Table 2. Secondary settling tank design criteria for activated sludge plants set out by the U.S. EPA (1975)

Type of process	Hydraulic loading (m/h)		Solidsloading ($kg/m^2/h$)		Sidewall depth (m)
	Average	Peak	Average	Peak	
Air activated sludge	0.68-1.36	1.70-2.04	4.08-6.08	10.17	3.66-4.57
Extended aeration	0.34-0.68	1.36	4.08-6.08	10.17	3.66-4.57
Oxygen activated sludge	0.68-1.36	1.70-2.04	5.08-7.13	10.17	3.66-4.57

Table 3. Optimum values of some design and operational parameters and unit sizes of secondary clarifiers (for 5000 m^3/d influent flow rate) according to WEF-ASCE design manual (1992)

Final settling tank	WEF-ASCE (1992)
Recycle ratio, %	43-60
SVI, ml/g	100 - 150
Sludge loading rate, $kg/m^2/d$	90 - 240
Overflow rate, $m^3/m^2/d$	16 - 48

Excessive solids in secondary clarifier effluents occur primarily because of one or more of the following (Ekama et al., 1997):

- hydraulic short-circuiting or resuspension of solids from the surface of the sludge blankets by high velocity currents;
- thickening overloads resulting in high sludge blankets and potential loss of solids when the blanket reaches the effluent weir;

- denitrification, causing solids to float to the surface;
- flocculation problems due to either flocs breakup or poor floc formation;
- insufficient capacity of the sludge collection system.

Summarizing, there are many factors that influence the clarification function of the secondary clarifiers. They include: flocculated state of the activated sludge, external dimensions of the clarifier such as surface area, depth, internal features such as inlet, outlet, sludge collection and baffling arrangement as well as flow disturbances (short circuiting, turbulence). In the last decade the designers have tried to incorporate an intentional flocculation zone into the secondary clarifier design to improve the effluent suspended solids concentration. In Europe such approach is rather rare though in the US this practice is becoming more and more common (Ekama et al., 1997).

Most of the clarification models reported in the literature are based on a statistical analysis of full- scale plant data, relating the effluent suspended solids concentration to a number of process parameters such as mixed liquor suspended solids, recycle flow rate, detention time, air flow rate in the biological reactor etc. Further description of the models of secondary clarifiers performance will be presented in chapter 5.

2.7. Thickening function in the sedimentation basin

In sanitary engineering thickening is usually thought of as a special treatment for sludge before its further treatment or disposal. However, thickening occurs together with clarification in any sedimentation operation and in many cases a satisfactory performance of a sedimentation basins requires production of both a clarified overflow and a concentrated underflow. Proper analysis of any sedimentation problem requires the determination of an area and volume for both clarification and thickening. The area required for thickening is determined by the solids transmitting capacity of the limiting concentration of the suspension. The larger of the two areas must be provided if the settling tank is to satisfactory perform both its clarifying and thickening role.

Thickening is defined as a sedimentation behavior characterized by line settling.

Solids subside with a clear line of division between settling solids and supernatant. Line settling assumes that all particles just below the solids-liquid interface have the same subsidence rate. Traditionally, thickening has been analyzed as though it occurred in two distinct steps. The first step is called zone settling and occurs when the solids concentration is sufficiently high to cause all the particles in the suspension, regardless of their size and shape, to settle at the same velocity. In this situation solids settle as a blanket and the velocity of subsidence being thought as an inverse function of concentration. As dilution is decreased the particles come closer together until at some point they become supported to some degree by particles directly below. Average settling velocity is retarded by the physical resistance of the particles below in addition to hydraulic considerations of water release. This step is called compression. Although it may be convenient to treat thickening as a two-stage operation it is doubtful that the two phases are actually as distinct as they are considered in the classical analysis of thickening.

Perhaps one of the most important developments in final clarifier theory was the realization of the importance of the thickening function and the estabilishment of solids loading as a critical factor in final clarifier performance (Dick, 1970; Dick, 1976). The reason for restricting the applied solids load on the secondary clarifiers is that each layer of sludge that might exist in the tank has some definite capacity for transmitting solids to the bottom of the tank (Dick, 1972). This capacity is determined both by the settling characteristics of the sludge and by the rate of removal of the return sludge. It is essential that the capacity of any layer that might exist in the tank for transmitting solids not be exceeded.The solids that cannot pass through the limiting layer will otherwise accumulate and in time move into the clarification portion of the setttling tank. Such an approach describing the solids thickening function of secondary clarifier is based on the flux theory and will be described in more details in chapter 5.1.

Sludge settleability has for long time not been featured directly into design criteria for settling tanks due to inadequacies in its determination (SVI). But it enters them in an indirect and empirical fashion, since for the various process types design criteria have been set down for proper functioning of the settling tank over a range of settling behavior. All these criteria accept that settleability will not fail below a certain minimum quality (see chapter 2.5).

Over the past twenty years research has been focused on developing improved measures for sludge settleability and on incorporating these into theories and design procedures. SVI and some modified indices of sludge settleability have been used for design and operation of secondary sedimentation basins. Two design procedures proposed by the ATV (Abwasser Technik Verband, Germany; 1973, 1976) and STOWa (Stichting Toestgepast Onderzoek Waterbeheer, Utrecht, The Netherlands; 1981) groups linked DSVI and sludge concentration to the permissible overflow surface loading found from experience, while SSVI test is associated with WRC (Water Research Centre) theory. These procedures allow calculation of the required secondary clarifier surface area (and for ATV also depth) to satisfy the thickening function of the clarifier for selected values of the sludge settleability. Although debate remains over the accuracy of these procedures they certainly present a significant step ahead, if compared with empirical hydraulic and solids loading rates that do not recognize sludge settleability and reactor concentration.

ATV relationship is based on the product of the biological reactor sludge concentration X (g/l) and DSVI (ml/g). The product is called the comparable sludge volume SV (ml/l) and ATV design criteria are valid for SV values less than 600 ml/l. Hydraulic surface loading qA (m/h) is calculated based on a permissible solids loading q_{00} and the comparable sludge volume SV:

$$qA = q_{00}/SV = q_{00}/ X *DSVI \qquad (8)$$

To maintain a low concentration of suspended solids in the effluent from horizontal clarifiers (SS < 20 mg/l) the permissible solids loading is q_{00} < 450 l/m^2 h (qA< 1.6 m/h). For vertical flow clarifiers and effluent SS <20 mg/l the permissible solids loading should not exceed 600 l/m^2 h (qA < 2.0 m/h).

The STOWa design procedure is a derivative of the ATV procedure (1981) since the authors found the flux and WRC procedures too inaccurate to accept them as a basis for design in the Netherlands. In their procedure the permissible overflow rate (q_A) is related the sludge volume DSV_{30} (which equals X *$DSVI_{30}$) and the sludge volume loading rate q_{SV} (which equals q_{A}*DSV_{30}):

$$q_A = 1/3 + 200/(X*DSVI_{30}) \quad (9)$$

for: $300 < q_A *X*DSVI < 400$ l/ m ^{2} h

and $q_A < 2.0$ m/h

If q_{SV} <300 or q_{SV} >400 l/m2 h it is set equal to 300 or 400 l/m2 h respectively and the overflow rate q_A is given by:

$$q_A = q_{SV}/DSV_{30} \quad (10)$$

These equations apply to both peak and average daily wastewater flows.

The Water Research Center procedure (White, 1975) is based on the flux theory for predicting the maximum solids handling capacity of settling tanks. The studies produced the following mass- flux equation:

$$G_p = 306.9(SSVI)^{0.77} Q_u^{0.68} / A^{0.68} \quad (11)$$

where. G_p - predicted solids loading rate, kg SS/m^2 h; SSVI - stirred sludge volume index at 3.5 g/l,ml/g; Q_u - recycle rate, m^3/h and A - surface area of the clarifier, m^2. As it was further investigated by Ekama and Marais (1986) this equation can be only used up to the critical underflow rate and they further developed the WRC procedure by providing additional relationship to calculate $q_{R\ crit.}$. as a function of SSVI.

The WRC procedure need to be used with certain constrains since it is developed based on SSVI and not directly on the zone settling rate of activated sludge. Research at full scale has indicated that the WRC procedure predicts within 20% the maximum solids handling capacity of a settling tank based on the SSVI as the sludge settleability measure (Ekama and Marais, 1986). The SSVI has been widely adopted in some countries as the routine sludge settleability measure and together with the WRC procedure is used for design and control of secondary clarifiers.

In cases in which maximum solids loadings are suggested values in the order of 98 kg/ day/m^2 are typical. Some ideas that have recently been suggested for using a single set of concepts to analyze the complete operation will be examined subsequently in the following chapters.

Currently the design of secondary clarifiers is usually done in two stages (Ekama et al. 1997): in the first zone settling and thickening considerations are applied that lead to the specification of a surface area and depth. With the proper incorporation of design features for clarification, zone settling and thickening criteria will govern the specification of the area of the clarifier. After specifying surface area and depth, the clarification efficiency of the tank is optimized by considering detailed design of the features such as: inlet arrangement, tank configuration, control of hydraulic flow patterns, short circuiting and turbulence with baffling, sludge transport and collection capacity.

Application of the solids flux procedure to a full scale rectangular and circular secondary clarifiers indicate that the flux procedure overpredicts the permissible solids loading by about 25% (Göhle et al. 1996;Ekama et al. 1997).

3. Process disturbances in the activated sludge process

The performance of the activated sludge process is limited by many factors. They are concerned with the biological activity of sludge microorganisms, hydraulic disturbances within the system that affect the ability of the clarifier to separate and concentrate the activated sludge from the effluent and with the sludge itself and its properties. Within this chapter the two first factors are only briefly disscused while the most attention will be focused on the sludge properties and its settling characteristics.

The amount of the solids and the particle size of the solids that do not settle are in some way related to how well the activated sludge is flocculated. If the effluent TSS is low enough the sludge is considered to be well flocculated. If the effluent TSS is high and the supernatant and effluent clear with the individually visible particles pinpoint flocs are being produced. If the effluent TSS is high and the supernatant and effluent are turbid deflocculation occurs. Despite several investigations of the flocculation of activated sludge and a number of different possible theories there is still not no generally accepted explanation of why activated sludge flocculates. Of the various theories that have been proposed to explain the bulking of activated sludge two are dominant: the filamentous theory and the bridging theory. Both of them will be further described in chapters 3.3. and 4.1.

Both bulking sludge as well as floating sludge are the phenomenon which are of much greater complexity than rising sludge or pinpoint floc, since the factors promoting the formation and development of a series of filamentous organisms responsible for the former have not been yet clearly determined. Both bulking sludge and pinpoint flocs tend to occur at low loadings with the highest SVI occurring at the extreme low end of the loading range. Although bulking and pinpoint flocs both occur at the low end of the loading rate they occur in different processes. Processes which tend to produce a pinpoint floc also produce sludge with a low SVI; processes which tend to produce bulking sludge do not produce pinpoint flocs.

Deflocculation can occur at any level of organic loading for variety of different causes; low dissolved oxygen, low pH and toxic shock loads are among the most frequent causes of deflocculation.

3.1. Sludge activity

Sludge activity expressed as sludge behavior and performance is very difficult to assess using measures of microbial biomass such as MLSS or MLVSS since it has not been possible to relate bacterial numbers directly to the process performance. The problems of estimating bacterial numbers, of differentiating viable and non-viable cells and of estimating bacterial levels of activity due to sludge age are difficult with the mixed population of activated sludge. So other methods were developed to assess the biological activity of sludge using chemical analyses rather then direct counting techniques. The methods assessing the biological activity of sludge comprise: measuring of adenosine triphosphate (ATP) which is used to measure the number of viable cells present in activated sludge, dehydrogenase activity (test TTC) or assimilation of glucose (Gray, 1990). By estimating the biological capacity of the activated sludge system, loading rates can be estimated based on the functional relationship between the concentration of wastewater and sludge activity.

Sludge biological activity can be affected by many different factors. Sludge activity is being inhibited by the presence of toxic materials (e.g. heavy metals). The toxicity of heavy metals in biological wastewater treatment processes depends on various process parameters (organic loading , sludge residence time, MLSS) and wastewater characteristics (heavy metals concentration, pH, form of metal).

According to Adams et al. (1975) the degree of toxicity of heavy metals is related to overall metal equilibrium in the aeration tank and is independent of heavy metals concentration, microorganism concentration and organic loading. The concentration of these variables affects the proportion of the active components tied up in the metal ligand complexes, and subsequently affects the extent of metal toxicity.

Tyagi et al. (1991) tried to develop a mathematical relationship between sludge solids concentration, recycle ratio and sludge settling velocity in relation to metal inhibition and demonstrated a rational analysis of final settling tank thickening performance. The results presented as an operational chart allow to determine for a particular reactor volume: (1) required values of MLSS and the recycle ratio for a given sludge age for different inhibitors concentrations, (2) flexibility of the plant to accommodate changes in the inhibitor concentration without altering the design, (3) the operating conditions with respect to inhibitor concentration that may lead to process failure and to estimate the variations in sludge age.

Clark and Forster (1983) has shown that the settling properties are related to biological activity of the sludge as measured by the concentration of adenosine triphosphste (ATP). The authors developed a mathematical model based on this hypothesis and tested it against the experimental data. The model was based on the concept that the SVI is the summation of the individual components of settlement indices and that these are dependent on the individual microbial activities as measured in terms of the ATP concentration.

By being able to estimate the biological capacity of the activated sludge system, loading rates including toxic substances can be based on the relationship between the concentration of the waste and sludge activity. This knowledge optimizes the process performance and reduces the possibility of overloading or inhibition.

3.2. Hydraulic disturbances

The problem of excessive hydraulic loading to a treatment facility, especially as a result of storm runoff often occurs at many wastewater treatment plants. Exceeding treatment capacity can jeopardize further treatment efficiency as well as result in serious effluent deterioration. Specially a secondary clarifier plays an important role in the overall process performance. Its hydraulic and solids handling capacities may actually dictate the effective treatment capacity of an activated sludge plant. Different countries have adopted different procedures to accommodate the peak flows.

Pumped flows can present a problem for secondary clarifiers in that square wave discharges may result; also size and features of the clarifier have a major influence on the effect of hydraulic transients (Ekama et al. 1997).

Monitoring of a sludge blanket level is very important to clarifier performance. A high sludge blanket renders a clarifier susceptible to sudden and critical effluent deterioration. Often, shock hydraulic loading do not of itself significantly affect solids separation efficiency. Other conditions such sludge as blanket level, sludge settleability and particularly a total hydraulic loading are far more important in determining what level of performance is achieved by the clarifier.

Typical designs for activated sludge systems do not ease the effects of hydraulic overloadings. However, the pilot studies conducted by Olsson et al. (1986) showed that the provision of freeboard in the aeration basin coupled with the use of an appropriate effluent weir design lessen the hydraulic effect and reduced the effluent SS. Equalization basins are occasionally used in wastewater treatment plants to minimize the effects of hydraulic peak flows on the biological treatment process.

Otterpohl and Freund (1992) developed a dynamic model for secondary clarifiers that would allow to predict settling of sludge flocs taking into account changing hydraulic conditions at dry and wet weather flows. The model predicts thickening and storage of sludge in the clarifier and estimates effluent concentrations of particulate substances at dry and wet weather flows.

3.3. Sludge properties

The ability of microorganisms to form flocs is vital for the activated sludge treatment of wastewater. The floc structure enables not only the adsorption of soluble substrates but also the adsorption of the colloidal matter and macro- molecules additionally found in most wastewater. The adsorptive capacity of flocs therefore facilitates the oxidation of this complex wastewater. However, important also is their ability to settle in a relatively short time under quiescent conditions; otherwise the biomass produced as a result of oxidation of the waste would pass to the receiving watercourse exerting a large pollution load.

There are four main phenomena that lead to a decrease of the quality of the effluent due to the escape of flocs:

- Bulking sludge due to an excessive growth of filamentous organisms and/or surface polymer presence (Sezgin et al. 1978, Jenkins et al. 1984, Chudoba et al. 1985; Forster, 1971; Pavoni et al. 1972)
- Pinpoint floc consisting of small floc particles present in the supernatant after the sludge has settled (Sezgin et al. 1978, Pipes, 1978, Palm and Jenkins, 1980)
- Floating sludge attributed to the presence of filamentous organisms that originate a foam in the aeration basin (Strom and Jenkins 1984, Pujol, 1991)
- Rising of the sludge after settlement in the clarifier due to the nitrogen produced in

the denitrification process (Wagner, 1984)
All of them will be discussed in the following chapters.

3.3.1. Sludge bulking

3.3.1.1. Filamentous bulking

<u>Filaments characteristics</u> When sludge settles poorly it is generally described as being a bulking type. Bulking is attributed to an infection of the sludge by filamentous organisms such as *Sphaerothilus natans*, Type 1701, *Nocardia* sp.Type 021N, Type 0041, Type 0092, *Microthrix parvicella* and others. Filamentous organisms extend from the floc particle, decrease its settling rate and physically hold the particles apart, thus preventing them from compacting to a high solids concentration. A various filamentous microorganisms can grow in different forms; rigid straight filaments that stick out into the bulking solution and coiled filaments that wrap around the flocs, the latter ones cause that a highly filamentous sludge can have a relatively low SVI. SVI of the filamentous sludge can change rapidly, whereas the SVI of a nonfilamentous sludge is stable (Pipes, 1979).

Filamentous microorganisms control methods require the proper identification of filaments because their appearance is associated with various cultivation conditions and technological parameters. Eikelboom (1977) as the first one developed a procedure for classifying the types of filamentous organisms based on their size, morphology and the response of the organism to standard staining technique. Out of about 30 different types of filamentous bacteria that have been distinguished one or more filamentous organisms are usually dominating in the particular activated sludge system. Since they have different growth demands and rates the factors affecting filamentous bulking can vary. The identification of the type of filamentous organism that predominates in the system can help to determine the causes and to identify potential solutions to the sludge bulking problem (Jenkins et al. 1986). The methods of microorganisms identification will be presented in more detailed way in chapter 4.3.

Many national surveys of filamentous organisms have been performed in different countries since Eikelboom's identification procedure. They include: Germany (Wagner, 1982) , France (Pujol et al. 1991), UK (Foot, 1992a), Australia (Blackall et al. 1991a), USA (Jenkins et al. 1986), Czech Republic (Wanner et al. 1998). They all tried to gain the knowledge on the distribution of filamentous microorganisms in individual countries together with the knowledge of finding proper control methods.

The results lead to the conclusions that there are many factors that influence the growth of filamentous bacteria. They include: low oxygen concentration, high oxygen concentration, high sulfide concentration, lack of certain nutrients like N and P, too high or too low sludge loading, high carbohydrate concentration, high concentration of fatty acids and the feed pattern. Some time bulking occurring in activated sludge processes at low organic loadings seems to involve the same filamentous organisms as bulking occurring at high organic loadings. This is consistent with the hypothesis that bulking is primarily a phenomenon occurring at low organic loading but can lead to high organic loading (and deflocculation) if it results in excessive solids loss or wasting (Pipes, 1979).

Bulking control The most frequent causes of filamentous bulking evoked by a wastewater composition and possible methods for its suppression and prevention are presented in Table. 4 (Chudoba, 1985). Non-specific bulking control measures (using chlorine, ozone or hydrogen peroxide) have been used to eliminate filament growth in the sludge, however those measures dealt with the symptoms of bulking i.e. they decreased the filaments but did not remove the causes of filament proliferation on a permanent basis. Therefore a specific control of bulking was needed that would identify and eliminate conditions that promote the proliferation of the specific filaments causing bulking problems.

A breakthrough in the control of bulking sludge was done by Chudoba (1973, 1974) and Rensink (1966, 1974). They concluded that the flow pattern in the reactor exerted an important influence on population dynamics and selection of various species in the mixed culture. The theory is based on the Monod equation presuming different growth constants μ and K_s for different species and consequently their different specific growth rates relationships to a substrate concentration. Complete-mixing reactors support the growth of filamentous microorganisms while reactors with a low degree of axial mixing i.e. with low dispersion number and high concentration gradients of substrate along the reactor suppress their growth. These phenomenon can be explained by the mechanism of filamentous organisms suppression and lower growth rate compared with floc formers at high substrate levels. The filamentous bacteria are more effective in their intake of substrates and oxygen at low substrate and oxygen concentrations due to their higher surface-to volume ratio while floc forming bacteria have higher growth rates at higher substrate concentrations. Many other experimental results confirm those findings (Wu et al. 1984; Novak et al. 1986), bacterial selection in this case is mainly due to different storage response to higher growth rate environment.

Table 4. The most frequent causes of filamentous bulking caused by wastewater composition and possible methods for its suppression and prevention (Chudoba, 1985)

Cause	Characteristic microorganisms	Suppression and prevention
Higher contents of sacharides	*S. natans*,Type 1701, Type 021N and others	Dosing of Cl_2 or H_2O_2 Application of selector
Higher contents of sulfides	*Thiothrix* sp., *Beggiatoa* sp.	Removal of S^{2-} by preaeration or precipitation
Nutrient deficiency (N,P)	*Thiotrix* sp. and others	Dosing of nutrients
Low pH	Fungi	Dosing of $Ca(OH)_2$

Horan and Shanmugan (1986) looked into effects of starvation and nutrient depletion on the settling properties of activated sludge. They found that during this period settleability, as determined by SSVI declined significantly. There was no change in sludge exopolysaccharide concentration or composition during the starvation period and loss of settling ability was attributed to extensive cell lysis, producing changes in floc structure from firm, round flocs to pin-point flocs. Appearance of filamentous bacteria during starvation suggested that these organisms had a growth rate advantage at low nutrient concentration.

Larrea et al. (1993) suggested that in mixed cultures cultivated with real wastewater a multiple limitation was quite probable (more than one type of substrate is limiting). It would explain bulking due to low dissolved oxygen concentration and/or low concentrations of some nutrients. In such case filamentous organisms can successfully compete with the floc formers even under high substrate concentrations because they are less susceptible to limitations by low DO or nutrient concentrations (low values of K_s for O_2 , N, etc.). To suppress the filamentous growth it would require to keep a high substrate concentration in the total volume of the aeration tank, which is not possible without a certain deterioration of the effluent quality. So other ways of maintaining a high substrate concentration are proposed as: selector, plug flow system, sequencing batch reactors.

Rensink and Donker (1991) confirmed that presence and configuration of the selector located ahead of the activated sludge tank played a decisive role in preventing or curing bulking sludge. A rectangular contact tank ensured well settling sludge while a circular selector (completely mixed) was not able to control bulking. Also great differences in biosorption and COD removal were measured between well-settling and poor settling sludge. Biosorption increased with increasing floc loading (up to the floc loading 60 mg COD/gMLSS) and with time.

Kinetic selection of the microorganisms in aerobic selectors may be an effective method for some types of filaments. As far as aerobic selectors are concerned several variables were proposed as a controlling factor: biochemical environment, soluble COD concentration, floc loading expressed as mass of applied soluble substrate per mass of sludge (mg COD/g VSS) and introduced by Eikelboom (1982), hydraulic retention time and dispersion number inside the selector. ATV Task Group on Bulking (1988) recommended for effective prevention of aerobic bulking aerobic selectors designed for sludge loading of 10 kg BOD_5/ m^3 d and anoxic selectors with sludge loading of 6 kg BOD_5/ m^3 d. Other selection design criteria include those suggested by Ramadori and Tandoi (1992):

Soluble COD concentration	$\leq$60 mg/l
organic loading	5-10 mg COD/mg VSS *d
Hydraulic retention time	5-30 min
Floc loading	~ 60 mg COD/g VSS

The selectors should be properly sized and compartmentalized. The selector should have four compartments and the concentration gradient of degradable substrate between the selector and the main aeration basin should be approximately 30 mg COD/l (Wanner, 1993).

Kruit et al. (1993) carried out the experiments to study the influence of O_2 in the selector on good setttling properties of the sludge and the substrate uptake in the selector. For this purpose a characterization of wastewater was done by fractioning COD according to the requirements of mathematical modeling of the activated sludge process (Henze et al. 1987). It was found that the presence of O_2 in the selector, at initial sludge loadings of 3.5-6.5 kg BOD/kg MLSS d, is important for producing a good settling properties of the sludge when the sum of readily biodegradable COD and rapid hydrolysable COD is greater then 40%. When the sum of sludge COD and slow hydrolysable COD is greater then 50% an unaerated selector can be used.

Since the biological selection of the floc forming microorganisms seems to be the most rational approach for control an excessive growth of filaments the other mechanism involved in bacteria selection has been investigated. It is called metabolic selection and is based on the different physiological adaptation of microbial cultures to rapid changes in growth environment (anoxic/anaerobic conditions). During such transient conditions the bacteria can use different sources of carbon and energy and different electron acceptors in biochemical reactions. They can also accumulate substrate and then convert it to storage compounds within the cells (Ramadori and Tandoi, 1992).

The metabolic selection under anaerobic and/or anoxic conditions has been demonstrated several times, even in completely mixed systems without any selective effect of the substrate gradient. Wanner et al. (1987a,b) reported that anoxic conditions were able to suppress the growth of Type O21N and *Sphaerotilus natans* since they could not use nitrite nitrogen as an electron acceptor. It was also found that severely filamentous mixed cultures had maximum rates of denitrification or nitrate respiration one order of the magnitude lower than non-filamentous mixed cultures. Also anaerobic conditions suppreses those species as a result of lower rates of polyphosphate depolymerization under anaerobic conditions.

In his later work Wanner and Novak (1990) demonstrated that filamentous bulking caused by so called oxic zone growers (e.g.O21N and *Sphaerotilus natans*) occurred when:

- a soluble substrate (already present or generated by anaerobic fermentation) penetrated from the anaerobic zone to the oxic zone as a result of the disability of poly P bacteria to sequester all this substrate under anaerobic conditions.
- a particulate substrate was transferred from the anaerobic to the oxic zone

enmeshed in the flocs of activated sludge, this phenomenon was caused by a rather low rate of utilization of high molecular soluble or particulate substrate under anaerobiosis.

In both cases the problems with filamentous bulking were more severe in the system with a completely mixed oxic zones then in the system with an oxic cascade.

Another issue were low F/M filaments that were not affected neither by selectors based on the preferential uptake of influent easily biodegradable COD by floc-formers through kinetic selection (aerobic , anoxic selectors) nor by metabolic selection (anaerobic reactors). Casey et al. (1993) looked into the hypothesis that the major factor influencing low F/M filament proliferation was intermittent aeration conditions (nitrification-denitrification). Because intermittent aeration causes organisms to be alternately exposed to aerobic and anoxic conditions it was proposed that a possible cause might lay in the requirement for the mass to switch between aerobic and anoxic metabolic pathways, this switching providing some competitive advantage to the filamentous organisms at the expense of the floc-forming organisms. The authors further explained the advantage as arising from the fact that the filaments reduced NO_3 to NO_2 only and were not inhibited in their oxygen uptake by nitric oxide, while floc-formers denitrified NO_3 to N_2 and accumulation of NO inhibited their oxygen utilization rate (OUR).

This theory was however questioned by Gabb (1993) who showed in his experiments that bulking activated sludge (with filamentous organisms present and DSVI> 200ml/g) reduced the nitrate/nitrite beyond just nitrite and that no inhibition of OUR took place. The full explanation of the causes and control of low F/M bulking still needs thorough evaluation and verification

Also sulfate reducing bacteria influence the population dynamics of the process. It was reported that some types of filamentous bacteria could utilize reduced sulfur as an energy source. Yamamoto et al. (1991) showed that when the sulfate reduction ratio increased Type 021N or *Beggiatoa* sp. grew in the sludge and settling characteristics became poor. These results suggested that sulfate reduction was a cause of growth of the filamentous bacteria which could utilize sulfide as the energy source. The results were confirmed by Echeverria et al. (1992) who stated that values of sulfide concentration in the influent above 1.0 mg/l negatively affected the sludge settleability causing sludge bulking and floating. Some sulfur microorganisms like *Thiothrix* exhibit the mixotrophy, i.e. in systems with anaerobic and oxic zones these organisms are able to utilize organic substrate in anaerobiosis by simultaneous heterotrophic reduction of sulfate sulfur. Reduced sulfur (S^{-2} ,S^{0}) is then oxidized autotrophically under oxic conditions to sulfates. Wanner et al. (1987b) showed that such a way of obtaining energy represented a metabolic advantage for some organisms, which in the case of *Thiothrix* might result in bulking problems.

At the present state of knowledge of non-filamentous organisms selection the most reliable way of suppressing filamentous bulking is to combine both the kinetics and the metabolic selection. Therefore the activated sludge with persistent bulking problem should be highly compartmentalized and flexible in order to switch from one to another type of zone. Anoxic and anaerobic selector system should be more effective in preventing filamentous bulking then aerobic selector, since filamentous bacteria (with some exception e.g. *Microthrix parvicella*) do not have the ability to carry out anaerobic respiration using nitrate as the electron acceptor or polyphosphates as a source of energy in order to accumulate organic carbon into cells. (Jenkins, 1993).

Summarizing their own experience and the available literature data Wanner and Grau (1989) concluded that the knowledge of a correct taxonomic position of filamentous microorganisms is helpful but not absolutely necessary for preventing or curing the bulking and foaming problems. They proposed a classification of filamentous organisms into four groups on the basis of their similarity, occurrence under the same operation conditions and causing the same problems. The groups are as follows:

- oxic zone growers S (*Sphaerotilus* - like microorganisms); they are composed of rod- or- square -shaped cells contained in a clear sheath and are able to utilize organic substrates only in oxic conditions. Their presence in activated sludges is connected with saccharidic or other readily biodegradable wastewaters, higher SRT and low DO. *Sphaerotilus* sp. are not capable of utilizing substrate under anaerobic and anoxic conditions; they have never been found in South African nutrient removal plants with anaerobic /anoxic zones.

- oxic zone growers C (*Cyanophycae*- like microorganisms, i.e. *Leucothrix, Thiothrix* and Type 021); the problem of identification of those species is very difficult. *Leucothrix* has not been mentioned in either Eikelboom's or Jenkin's Manual (Jenkins et al. 1986) since it is considered to be a of marine origin (Brock, 1966). Both the genus *Leucothrix* and Type 021N are connected with the filamentous bulking caused by readily biodegradable wastewaters, high SRT and probably by nutrient deficiency. There is no evidence that they could cause bulking problems in the system with anaerobic/anoxic zones. Contrary to oxic zone growers S the microorganisms from group C are able to metabolize sulfur (excluding *Leucothrix*). Especially sulfide oxidizing bacteria such as *Thiothrix* can cause serious bulking problems whenever it can use its mixotrophic abilities (Nielsen, 1984, Wanner et al., 1987 b,c)

- all zone growers A (*Mictrothrix parvicella* and other heterotrophic microorganisms capable of utilizing substrate not only under oxic conditions); they are connected with low F/M and are observed in nutrient removing wastewater treatment plants. Interest in *M. parvicella* has increased when it was found that it has ability to utilize the substrate in anoxic condition and also can accumulate some degradable substrate. This way it can compete against floc-forming bacteria (Slijkhuis and Deinema, 1982; Wanner and Grau, 1988). Such ability may explain the fact that selectors do not control SVI to low levels when *M. parvicella* exists. The survey prepared by Blackbeard et al. (1988) suggests that Type 0092 is the microorganism the most frequently found in activated sludge from nutrient removal plants; another all zone grower is *Nostocoida limicola* (Wanner and Grau, 1989). All these organisms may be generally classified as organisms which are able to accumulate or utilize substrate under anoxic or anaerobic (polyphosphate formation and degradation) conditions.

- foam-forming microorganisms F (*Noccardia spp.*, *Nostocoida limicola,* Type 0092 and other microorganisms producing biosurfactants). These type of microorganisms will be discussed in more detail in the chapter on foaming.

<u>*Bulking problems in nutrient removal plants*</u> - Whereas the principle and mechanism of suppressing the growth of filamentous microorganisms in conventional activated sludge systems are well understood and were successfully applied in practice, the explanation and control of filamentous bulking in nutrient removal plants is still difficult. In an activated sludge system modified for biological nutrient removal, filamentous microorganisms can utilize a substrate either under anaerobic or anoxic conditions or under oxic conditions. Particulate organic substrate, present in high quantity in domestic sewage, is firstly enmeshed in the sludge and then hydrolyzed to a readily biodegradable soluble COD by hydrolytic enzymes. This process occurs

mainly in aerobic zone; here the hydrolysis products become available for microorganisms at very low concentration, thus favoring the growth of filamentous bacteria (Ekama and Marais, 1986). Apart from low F/M conditions bulking in nutrient removal systems may be induced also by: unaerated mass fraction, frequency of alternation between anoxic and aerobic conditions, low DO concentration, anoxic or aerobic reactor nitrate and nitrite concentrations ect.

The most extensive survey of filamentous population in nutrient removal plants in Denmark, Greece, Germany and Netherlands have been conducted recently by Eikelboom (1998). They found out that biological nutrient removal processes favoured filamentous organisms in their competition with floc forming organisms. An increase in the size of the filamentous population resulted in a deterioration of the settleability, except for plants with Bio-P removal conditions. The authors assumed that the dense clusters of Bio-P bacteria increased the weight of the flocs, and compensate for the large number of filaments. *M. parvicella* was found to be the most important filamentous species in nutrient removal plants. It seems to proliferate if the particulate fraction is first hydrolyzed or if it enters the plant via an oxic zone. It is concluded that bulking in nutrient removal plants was mainly caused by filamentous species requiring the particulate fraction for their growth.

M. parvicella species have stirred a vivid discussion among the researchers. There is still much controversy on the subject of its physiological properties and different performance in pure cultures isolated by the different researchers and within the activated sludge flocs. Evidence for effectiveness of selectors in controlling *M. parvicella* is ambiguous too. Cases were reported when *M. parvicella* caused serious bulking problems in the systems with anaerobic zones and it seems that it may be one of the most common filamentous microorganisms in nutrient removal plants (Mulder and Rensink, 1987; Rensink and Donker, 1987; Blackbeard et al. 1988). There are reports in the literature that claim effective *M. parvicella* control with use of anaerobic selectors (Daigger et al. 1985), on the other hand recent studies indicate that selectors are unsuccessful in controling low F/M bulking (Mamais et al. 1998). According to the latter studies the best control strategy for *M. parvicella* is adoption of continuous plug flow reactors, due to the following advantages: utilization of the higher sorption capacity of the floc formers to remove greater amount of slowly biodegradable COD and avoidance of the dispersion of the soluble products of colloidal material hydrolysis. Knoop and Kunst (1998) investigated the influence of low (<0.1 kg/kg d) BOD sludge loading rates and low temperatures on the growth of *M. parvicella* and found that the optimum growth of these species is induced at temperatures <12 to 15^{0}C and sludge loading rates of >0.1 kg/kg d.

Modeling of filamentous sludge bulking Some researchers tried to model filamentous activated sludge and the relationship between filamentous and floc-forming bacteria. Lau et al. (1984) tried to model filamentous activated sludge bulking attributable to low DO concentrations and to explain the relationships between organic loadings and DO concentrations required to prevent bulking. They developed a mathematical model incorporating simultaneous diffusion of carbonaceous substrate (glucose) and DO through a floc and accompanying growth of two microorganisms (floc former - *Citrobacter sp.* and filamentous bacteria *S. natans*). It was found that low bulk DO concentration, high bulk glucose concentration, large floc sizes and spherical floc shapes tended to favor *S. natans*. With increased bulk glucose concentration (increased organic loading) the model predicts that a higher bulk DO concentration is required to prevent bulking.

Then Chiesa and Irvine (1985) introduced ”integrated hypothesis” centered around 3 model microorganisms (floc formers, slowly growing, starvation resistant filamentous organisms and rapidly growing, starvation susceptible filamentous organism) to explain the growth and control of filamentous organisms using combined kinetic selection and accumulation/storage concept. The model organisms were characterized by: maximum growth rate, half saturation coefficients for carbon substrate and DO and endogenous decay rates. The authors first introduced the kinetic properties and metabolic abilities of the filamentous organisms into a process modeling.

Kappeler and Gujer (1994) developed the AEROFIL model to describe the competition between facultative aerobic floc-forming, obligate aerobic filamentous and nitrifying microorganisms in the case of aerobic bulking and they introduced typical kinetic parameters and wastewater fractions. In the model it was hypothesized that the description of the competition between floc-forming bacteria and filamentous organisms requires the distinction of readily biodegradable substrate from the influent and readily biodegradable hydrolysis products, due to the different diffusional resistance. Most of the kinetic parameters of the heterotrophic floc-forming and the nitrifying microorganisms can directly be estimated from different batch-tests. Because obligate aerobic filamentous microorganisms only represent a small fraction of the biocenosies, their kinetic parameters cannot directly be measured, but must be estimated with indirect methods.

The validity of the AEROFIL model to predict the biocenosis composition is restricted to: (1) completely aerobic systems with solids retention time less then 10 days; (2) aerobic systems with a small anoxic zone and solids retention time less then 10 days; (3) aerobic systems with a small anaerobic zone and no biological phosphorus removal - due to reduced solids retention time there is no nitrification; (4) no relevant amounts of surfactants which might favour the proliferation of Actinomycetes, or long chain fatty acids which might favour the proliferation of *Microthrix parvicella,* neither in the influent nor produced in the wastewater treatment plant itself. If these models limitations are exceeded the most probable biocenosis composition cannot be predicted any more, because other groups of filamentous microorganisms such as *Actinomycetes* or *Microthrix parvicella* may become important.

The model was further verified by Kappeler and Gujer (1994a) during experiments in a pilot- and full-scale. The prediction of the AEROFIL model lead to COD concentrations of facultative aerobic floc-forming, obligate aerobic filamentous and nitrifying microorganisms in the activated sludge. Until now it is not possible in practice to determine the COD concentration of the three groups of microorganisms separately within the activated sludge. Therefore the model‘s verification was carried out qualitatively by comparing the observed sludge volume indices with the predicted fraction of filamentous microorganism in the activated sludge. It was shown that long hydraulic retention time in primary settling tanks tend to favour aerobic bulking. The design of aerobic selectors with 10 kg BOD_5/m^3 d, as recommended by the ATV Task Group on Bulking (1998), generally leads to optimal selector volumes. The positive effect of compartmentalized aerobic selectors is demonstrated by the AEROFIL model. If anoxic selectors are designed with 2.5 kg BOD_5/m^3 d, as recommended by ATV Task Group on Bulking, the growth of obligate aerobic filamentous microorganisms is successfully suppressed. Extended anoxic sludge blankets in secondary clarifiers hinder the obligate aerobic filamentous microorganisms in their competition against facultative aerobic floc-formers.

Casey et al. (1994) hypothesized that low F/M bulking and scumming is caused by increased concentrations of nitric oxide, leading to inhibition of heterotrophic facultative aerobic floc- forming microorganisms in aerobic zones. Because of the lack of at least some of the reductates necessary for denitrification, the inhibitory denitrification intermediate nitric oxide is not built up in the filamentous microorganisms. This hypothesis might be a key explanation in the low F/M bulking and scumming.

Based on this hypothesis Kappeler and Brodmann (1995) formulated a mathematical simulation model which predicts the most important observation related to low F/M bulking and scumming: surfactants and hydrophobic substrates may lead to the proliferation of low F/M filaments and low F/M bulking and scumming is normally not suppressed by aerobic or small anoxic selectors. Beside these well established facts, the model predicts that: (1) incomplete nitrification can cause serious low F/M bulking and scumming problems, (2) large anoxic selectors with complete denitrification prevent low F/M bulking and scumming (3) extended sludge blankets in the secondary clarifier influence the proliferation of low F/M filaments; in completely aerobic systems in a negative way, in systems with predenitrification zones in a positive way.

Another approach was presented by Takacs and Fleit (1995). They tried to merge the morphological theories with physiological (metabolic) characteristics of the activated sludge bacteria and by this way explain filamentous bulking. The authors developed a dynamic mathematical model to simulate the population dynamics of floc-formers and filaments within the activated sludge floc. The concentration of DO and soluble substrate was calculated inside the floc core under different bulk concentration conditions. Dynamic simulations runs were performed to calculate the growth of the two morphological types of microorganisms inside the floc under diffusion governed conditions. The results indicated that the model accurately predicts the onset of excessive filamentous growth (directly linked to bulking) even when traditional models neglecting diffusion limitation fail to do so. The authors also demonstrated the positive feedback effect of the unidirectional growth on the selective enrichment of filamentous organisms under low DO or F/M conditions.

There are two major obstacles that make it difficult to incorporate these and other factors influencing activated sludge settleability into an integrated model (Scuras et al. 1998). First, the correlations currently relating each one to the settleability only apply to the specific conditions under which they were determined. Second, according to the IAWQ's Activated Sludge Model models of floc former/filament competition and extracellular polymer production often relate their results to substrate concentration in the bulk liquid of the reactor while these concentrations vary within the microenvironment of the activated sludge floc. Scuras et al. (1998) have presented the approach to modeling the activated sludge microenvironment and its effects on the settleability through population dynamics. Their particular interest was focused on the microcolony porosity. A model was developed to explore the relationship between biomass growth, death, and hydrolysis; soluble component diffusion and microcolony porosity.

<u>Filamentous bridging theory</u> Filamentous bacteria apart from being a nuisance in an activated sludge process operation are considered an important contributor in the floc forming process. Lau et al. (1984) have looked more closely into a filamentous theory of floc forming. They found that the formation of aggregates similar to those of activated sludge only occurred during the period when both organisms floc-forming bacteria (*Citrobacter sp.*) and filamentous bacteria (*S. natans*) were present in culture. Whereas, if the two organisms were grown separately only small compact flocs (*Citrobacter sp.*), short filaments (*S. natans*) or single cells could be found in the pure culture. Also Wanner et al. (1998) in their national survey of activated sludge separation problems in the Czech Republic discovered that the relationship between the abundance filaments and settling properties expressed in terms of SVI was rather loose. Low SVIs were related more with good activated sludge floc properties then with the absence of filaments. On the other hand, a very high abundance of filaments always resulted in bulking or foaming. More on the filament floc formation theory is decribed in chapter 3.1.

3. 3. 1. 2. Zoogleal bulking

The activated sludge bulking can also occur when filamentous microorganisms are completely missing. Such phenomena is called zoogleal or more exactly non-filamentous bulking and is related to viscous exocellular polymers produced in excess by some kinds of bacteria (i.e. *Zooglea sp., Acinetobacter sp.).* The exocellular slime capsule absorbs a lot of water that results in a voluminous character of the sludge flocs. In such a case the sludge exhibits high sludge indices (SVI) and loses the ability to flow. This phenomena was for the first time described by Heukelekian and Weisberg (1956).

Mueller et al. (1967) working with the pure culture of *Zooglea ramigera* strain I-16-M which is considered a floc forming bacteria found that its flocs were as large as 1150 μm in length or 260 μm in width; it indicates that large flocs could form even without the existence of filamentous organisms.

Novak et al. (1993) summarized several different reasons of non-filamentous bulking. They include:

- the composition of wastewater supports the formation of zoogleal growth, mainly the high content of low and long chain fatty acids and oleic compounds;
- the high sludge loading and/or an influent containing an insufficient amount of certain nutrients;
- in some cases selector systems significantly stimulate the growth of zoogleal colonies;
- in anaerobic and/or anoxic systems which biologically remove phosphorus, an excessive growth of poly-P bacteria , e.g. *Acinetobacter sp*. which form clusters of bacteria glued together by ECP can cause some settling problems;

The authors investigated a case of zoogleal bulking appearance in a bench -scale activated sludge plant with nitrification and denitrification. The bulking period (SVI over 300 ml/g) was observed during the experiment due to the occurrence of viscous jelly-like activated sludge. The significant production of exocellular polymers was related to the change of wastewater composition. When oleic acid was used instead of lecithin the non-filamentous bulking was accompanied by an intensive foam production in the system. When the wastewater returned to their original composition, biocenosis of the bulking and foaming sludge gradually changed and after 3 weeks the vicious zoogleal colonies were replaced by compact well-settling flocs.

Still, some researchers suggest that many older reports of non-filamentous bulking probably are the result of other settling problems, such as deflocculation, pin point floc or dispersed growth rather then true bulking. Strom and Jenkins (1984) pointed out that some of the filaments, particularly *H. hydrossis*,are very narrow and difficult to see; they may be easily overlooked particularly if the observer is looking for the much larger *S. natans.* That may provide another explanation for some cases of non-filamentous bulking. True *Zooglea* floc with a distinctive type of fingerlike branching were seen only rarely and never as a predominant form. Thus zoogleal bulking must be rare now though it may have been common at one time and in practice it does not bring a considerable operational problem.

3.3.2. Dispersed sludge (pin-point sludge)

It was observed long ago that sludge experiencing an environmental shock would undergo an observable change in settling abilities. Under such conditions activated sludge microorganisms do not agglomerate and form a settleable flocs, but occur mostly in the form of non- settleable suspended solids (pin-point floc). These dispersed biosolids (primary solids) either have not been incorporated into a floc particle or have been sheared from a floc particle as a result of excessive turbulence in the aeration basin or some other operational disturbances. Secondary effluents from such systems are very turbid and have high values of BOD and COD. Total dispersed growth is very rare in the wastewater treatment practice; usually part of the mixed culture forms well separable flocs and the other part is present in non-settleable dispersion.

The causes of the dispersed growth can be divided into two groups (Chudoba, 1989). One group involves parameters of the technological process e.g. MLSS concentration, sludge retention time (SRT). Usually, lower SRT results in higher proportion of sludge becoming a non-settleable dispersion. At such a high loading bacteria live in the environment containing higher concentrations of substrate, and so they cease producing glycocalyx which is responsible for flocculation. The other group involves factors connected with the wastewater composition. Temporary deflocculation also occurs as a result of the sudden changes in temperature, salinity and pH value (Pipes, 1969). Permanent deflocculation may occur due of the absence of macro or micronutrients or the presence of toxic materials in the process; the toxicity of heavy metals results from the binding reactions between them and the active cellular components to form inactive complexes (Pearson and Dugan 1971).

The pinpoint floc occurs usually as a result of dispersed growth e.g. insufficient production of glycocalyx due to a low organic loading, however there are also other reasons determining its presence in the activated sludge system (Eckenfelder and Grau, 1992):

- absolute absence of filamentous organisms
- disintegration by the shearing effect (mechanical aerators)

Neufeld (1976) has followed the heavy metals-induced deflocculation of activated sludge. The metals considered were mercury, cadmium and zinc and they were added to the wastewater in concentrations of 0, 30,100, 300, and 1000 mg/l. In his experiments he found that the influence of the shock loading of heavy metals on activated sludge settling may seem initially to have a beneficial effects based on the observation of reduced SVI at the higher metals loading but not all the sludge settle. However, at the same time shock loadings of heavy metals resulted in the formation of a highly stable pin-point floc. This condition termed" sludge deflocculation" resulted in the loss of significant quantities of biomass.

3.3.3. Foaming

The types of foams can be classified according to the conditions under which they occur:

3.3.3.1. Denitrification foaming

In plants performing a denitrification process the release of nitrogen microbubbles occurs which lower the apparent density of sludge and favours the flotation particularly in the secondary clarifier. The phenomenon of the sludge rising due to denitrification has been reported in the literature early as 1940s (Sawyer and Bradney, 1945). Since the critical amount of N_2 gas required to cause sludge rising depends on many factors e.g. denitrifier fraction, biomass concentration, the amount of adsorbed slowly biodegradable organics and nitrate available for endogenous nitrate respiration (ENR) sludge rising problems may occur under different operational conditions.

Studies conducted by Clayfield (1974) and Henze et al. (1993) on the sludge rising, pointed out that the most important parameter is the rate of ENR (r_{ENR}) which depends on the specific ENR rate constant k_{ENR} (1/d) and the biomass concentration

X (mg/l) as follows:

$$r_{ENR} = k_{ENR}\,X \qquad (12)$$

For sludge to become unstable a sufficient amount of N_2 gas must be produced through ENR.

Henze et. al. (1993) discussed the factors influencing the actual concentration of nitrogen gas at a given level in the settling tank. They included: solubility of nitrogen gas, depth below water level, nitrogen gas concentration in settling tank influent, nitrogen gas production rate, oxygen concentration in the settling tank influent, water passage time from settling tank inlet to its final destination within the tank and nitrate available for nitrification. The most important factor is the rate of biological denitrification. Factors like nitrogen gas solubility and oxygen concentration in settling tanks are less important. The authors stated that for all wastewaters the nitrate concentration in the effluent from the nitrifying treatment plant will be above the one critical for rising sludge, which at 20^0 C is around 6-8 g $N0_3$-N/m^3 .

Clayfield (1974) presented an equation to estimate the amount of N_2 gas (W) necessary to render a sludge blanket unstable as follows:

$$W = \left[(0.019p - 0.015) + \frac{Xp}{858d_w}\left(1 - \frac{d_w}{d_s}\right)\right] / 1000 \qquad (13)$$

where X - biomass concentration (mg/l); p -pressure (atm); d_w - density of water (g/cm^3); d_s - sludge density (g/cm^3). The total amount of N_2 gas necessary for sludge rising increases with depth. However, the rate of ENR also increases with depth due to higher solids concentration, thus providing more N_2 at higher depths.

In order to compare those two opposing factors for sludge rising, a sludge stability index SSI (gNO_3/gN_2) or the ratio of amount of nitrate utilized through ENR to that amount necessary for sludge rising was defined by Kim et al. (1994) to determine whether the particular sludge layer is unstable:

$$SSI = \frac{C_0 - C(z,t)}{W(z,t)} \quad (14)$$

where z - sludge depth, cm; t- time, min; C- nitrate concentration, mg/l; W - total nitrogen gas necessar to rise sludge, mgN/l.
The SSI value above 1g/g indicated that a particular sludge layer is unstable. Apart from SSI Kim et al. (1994) have developed a comprehensive model based on an extension of the solids flux theory to account for biomass stratification during sludge settling and corresponding differences in r_{ENR} values. Results from laboratory-scale batch settling experiments were used to verify the model. The model predicts both the solids and NO_3 profiles reasonably well.

The problems of rising and floating sludge associated with degasification in secondary clarifiers can be minimized by implementing and optimizing denitrification in all nitrifying activated sludge processes, even those not required to denitrify (Ekama et al. 1997). In addition to minimizing the concentration of nitrate entering the secondary clarifier, the flotation problems can be minimized through changes in clarifier design and operation (increase of the sludge recirculation ratio).

3.3.3.2. Bacterial foaming

<u>Microbial characteristics</u> The activated sludge foaming was initially connected with the presence of branching actinomycete bacteria *Nocardia amarae* (Lechevalier and Lechevalier, 1974) which produces a lipid material which is excreted into the mixed liquor. The lipid material collects on the surface of air bubbles. The bubbles mesh together including some of the acinomycetes colonies and float to the surface to form scum (Pipes, 1978). Foam density of about 0.7 (Jenkins et al. 1986) is sufficient to prevent wind from blowing the foam away. Within the floating sludge blanket actinomycetes have several advantages over other sludge bacteria. They are resistant to dryness because of their waxy cell walls, they are able to store poly -β -hydroxybutyric acid (PHB) and polyphosphates and thus to stand starvation. Therefore actinomycetes do not only survive but even grow in the floating sludge.

However, the flotation of a fraction of the sludge is also related to the metabolism of certain filamentous microorganisms which synthesize and /or excrete hydrophobic compounds (Pitman, 1984). The most common organisms that have been identified include *Rhodococcus rhodochrous* (Lemmer and Kroppenstdt, 1984) and long unbranched filamentous *Microthrix parvicella* (Jenkins et al. 1984) The origin of the development of such filaments are complex; at present stage of knowledge the most important ones seem to be: nutritional conditions, introduction of hydrophobic substances (greases, oil) into wastewater, certain mechanical aspects (aeration, stirring).

Goddard and Foster (1987) reported that also *Nostocoida limicola* and Type 0041 cause foaming since both species produce biosurfactants (lipids, lopipeptides, proteins) that form a foam. Both *N.limicola* and *Type 0041* were identified in the activated sludge from the biological phosphorus removal plant (Wanner and Grau, 1989). According to Blackbeard et al. (1986,1988) the most frequently observed microorganisms in the foam from nutrient removal activated sludge plants in South Africa are: Type 0092, *M. parvicella* and Type 0041, while *Nocardia* occurs less frequently. Recently, a branching nocardiaform, *Nocardia pinensis* was added to the list of foam causing actinomycetes detected in Australia (Blackall et al. 1989).

Because the first branching acinomycete isolated from activated sludge foam was *Nocardia amarae* these foams are commonly refered to as Nocardia foams despite the isolation of other genera. Since many of these have similar branching morphology they are sometimes called NALOSs (*Nocardia amarae* - like organisms) to distinguish them from another foam - former *Nocardia pinensis* (Blackall et al. 1989a) which has typical tree-like morphology. Goodfellow et al. (1994) proposed to transfer and reclassify *Nocardia amarae* to the genus *Gordona* and since then the name *Gordona amarae* and GALOs (instead of NALOs) has been introduced for these species.

Lemmer and Baumann (1988) looked into types of synergism between different types of sludge bacteria. Influence of *Pseudomonas sp.*, *Alcaligenes sp.*, *Flavobacterium* sp. and *Acinetobacter sp.* on acinomycetes growth and biomass production of two-strains cultures was investigated. It was found that *N. amarae* growing together with *Acinetobacter sp*. showed significantly higher biomass production than the sum of the biomass production of the controls. No such positive influence was detected for the other pairs tested. It is known that both actinomycetes and *Acinetobacter* can store poly-β- hydroxibutyric acid when oxygen is plentiful and thus survive limited anaerobic conditions better then other aerobic sludge.

Blackall et al. (1991) have studied strains of *Nocardia amarae* to evaluate their kinetics coefficients of growth, organic substrate and oxygen uptake and nitrate use. These data were compared with those for other activated sludge organisms (*Zooglea ramigera,* Type 021N and other floc formers) to evaluate the potential influence of various plant design and operating modifications. The authors reported that *Nocardia* was a poor competitor with both activated sludge floc formers and bulking filamentous organisms for acetate and oxygen; it did not take up acetate under anoxic and anaerobic conditions and denitrified slowly. These observation suggest that properly designed anoxic and anaerobic selectors would be effective in controlling *Nocardia* in activated sludge.

Kocianova et al. (1992) studied foam formation and confirmed that *M. parvicella* can exhibit a variable morphology. The physicochemical and biological examination of activated sludges showed that *M. parvicella* as well as altering its morphology can exist in both as hydrophilic and hydrophobic particles depending on the nutrition conditions.

Chacin et al. (1994) examined foams generated in anaerobic sludge digesters. They have found that *M.* p*arvicella,* the most common foam-forming species in activated sludge plants in the UK, is associated with the foams produced in anaerobic digesters and this organism can grow in a strictly anaerobic environment. However, the growth did not take the normal form of long filaments; it rather was seen as shortened filaments. It can be hypothesized that many of filamentous microorganisms found in activated sludge foams may be only „ entrapped" into the foam formed by the true foam-forming microorganisms.

<u>Foaming control</u> The most successful method of preventing the *Nocardia* growth is lowering the SRT since *Nocardia* spp. are slow growing organisms and at the high SRT they have a metabolic advantage in competing for substrate under low F/M conditions. This method however cannot be applied at the nutrient removal plants since it is in contradiction with the requirements of the nitrifiers and they are being washed out from the system. When nitrification is required properly design anoxic selectors would be effective in controlling *Nocardia* growth (Jenkins et al. 1984; Pitt

and Jenkins, 1990; Blackall et al. 1991). Another solution may be selective foam wasting that uses the increased aeration to strip the foaming organisms from the MLSS into the foam and then selectively wasting the foam (Richards et al. 1990). Lowering SRT is also not successful for all foams caused by actinomycetes. They can be suppressed by decreasing the air flow rate, pH value and placing the selector in front of the aeration tank.

Control of biological foaming, besides the emergency modifications of usual plant operation (e.g. aeration, recirculation, sludge withdrawal) can be undertaken as the actions presented in the following table.

Table 5. Effectiveness of the technical solutions (Pujol et al., 1991)

	Successful in % of cases	Number of cases
Mixing zone	73	11
Oxidant addition	66	9
Anti-foaming addition	57	7
Coagulant addition	46	28

As in the case of bulking only an approach based on microscopic follow-up examination of sludge and the good knowledge of a specific plant can guarantee the efficiency of the technical solution.

3.3.3.3. Surfactant foaming

Foams can be also chemical in nature; they are usually not so persistent and difficult to remove as biological ones. Large quantities of white frothy foam are often generated on the surface of the aeration tanks and clarifiers during start-up of the plant. This material is probably the accumulation of undegraded surface active organic matter and usually disappears once the sludge mass becomes established. More persistent foams are common in the plants were massive use of even biodegradable detergents and heavy inflow of colloidal organic matter or hydrocarbons occurs. If such an inflow remains occasional the foaming process may affect the plant for a short time only, or it may persist and in the long run cause the development of stable biological foam (Pujol et al. 1991).

Non - ionic synthetic surfactants are used widely in commercial and industrial cleansing applications. They are commonly present in the US wastewaters in concentrations ranging from 1 to 20 mg/l (Ho and Jenkins, 1991). Because the surfactants have ability to lower surface tension and thereby stabilize the liquid film between air bubbles it is possible that their presence in the activated sludge containing *Nocardia sp*. could stabilize the foam of *Nocardia sp*.

Ho and Jenkins (1991) conducted research work to investigate the influence of surfactants on *Nocardia sp*. foaming. They discovered that the foaming of activated sludge was significantly enhanced by the presence of non- ionic surfactants, but surfactants alone could not generate a stable foam if the sludge did not contain *Nocardia* cells. The foaming of *Nocardia* containing activated sludge transported suspended solids and *Nocardia sp*. filaments into the foam and increased their level over those found in the mixed liquor.

Also overdosing of polymers in sludge dewatering equipment has been indicated as a possible cause of scum formation (Bradley and Kharkar, 1996).

4. Sludge characterization

The overall efficiency of a biological treatment process depends on the settling characteristics of the activated sludge flocs. Numerous investigators have reported on the settling characteristics and these reports can generally be grouped into the following categories: (1) the relationship between the settling characteristics and the naturally produced polyelectrolyte which forms a bridge between individual flocs (2) the relation of the settling characteristics to electrochemical potential properties of the floc that has been considered as stabilization of colloidal particles and (3) its relation to physical properties such as the size and density of the floc.

Activated sludge flocs have a very complex structure and there are various techniques available to describe their physico-chemical structure. Urbain et al. (1993) have made a summary of the most common methods to describe floc characteristics. The various factors normally studied include: amount of filamentous microorganisms, floc size, specific surface area, surface charge, amount of extracellular polymers, amount of divalent cations (e.g. Ca^{2+}, Mg^{2+}), floc strength, floc density and hydrophobicity. Barber and Veenstra (1986) found filament length to be single the most important factor among volatile fraction, bound water content, electrophoretic mobility, filament length and quantity and floc particle size and quantity in gravitational thickening.

All these important factors influencing the activated sludge settling characteristics are described in more details in the following discussion.

4.1. Biochemical characterization

4.1.1. Exocellular polymers (ECP)

4.1.1.1. Exocellular polymers vs. filamentous bridging theory

Microbial aggregates encountered in wastewater treatment systems, such as activated sludge flocs and biological films are composed of bacterial cells, extracellular polymers water and particulate matter retained from the treated wastewater. Some researchers (Busch and Stumm, 1968, Pavoni et al. 1972, Chudoba, 1989) question the theory of the filament backbone structure of the flocs and suggest that apart from filamentous growth also flocculation and the types of exocellular polymer present at the sludge surface are factors related to sludge settlement. Busch and Stumm (1968) reported that dosing the extracted polymer from well-flocculated activated sludge into poorly-flocculated sludge shows a remarkable effect on improving the flocculation.

The surface of sludge microorganisms and the sludge matrix are important in the flocculation process. Even accepting the filamentous concept, Pipes (1979) suggested that the surface charge may be a deciding factor where bulking is concerned, since it will influence whether filaments either protrude and hold particles apart or coil around them to permit a close packing. The importance of the nature of the surface of the sludge flocs has stimulated research into the nature of exocellular polymers produced by bacteria and their significance in flocculation.

Both models- polymer bridging (Busch and Stumm, 1968; Pavoni et al. 1972) and filament backbone model of Sezgin et al. (1978) share certain features in common, for instance the role of the polysaccharides in the maintenance of the sludge surface charge (Forster and Dallas-Newton, 1980) but disagree on the origin of the forces required to bind together the floc constituents. The polymer bridging model accepts the electrostatic bridging of microorganisms onto a large three-dimensional matrix with ionogenic groups being provided by high molecular weight polymers such as polysaccharides found at microbial surfaces, either in the form of an attached capsule or a discrete slime layer. Foster (1971) who measured the electrophoretic mobility of sludge flocs postulated the most important polymeric material was polysaccharide, with the principal ionogenic monomer being glucuronic acid which would be fully ionized at pH -values of activated sludge.

The filament backbone model suggests that floc rigidity is provided by filamentous bacteria which form a backbone around which exopolysaccharide producing organisms such as *Zooglea* attach and form a gelatinous matrix to which colloidal material and other organisms adhere. Experiments of nutrient starvation conducted by Horan and Shanmugan (1986) caused a rapid utilization of the intracellular storage polymers (PHB) and decline in sludge respiration rate. During this period sludge settleability, as measured by SSVI and ease of dewatering, as determined by specific resistance to filtration declined markedly. There was no change in sludge exopolysaccharide concentration or composition and loss of settling ability was attributed to extensive cell lysis, producing changes in floc structure from firm, round flocs to pin-point flocs.

Another forces and mechanisms involved in the flocculation include:

- double layer theory (Zita and Hermanson, 1994, used potassium and calcium ions to show that flocculation dependent on double layer theory rather then ion bridging mechanism).
- hydrophobic interactions (Urbain et al. 1993 introduced a ”hydrophobic effect” that resulted from the behaviour of particles or molecules incapable of interacting electrostatically or establish hydrogen bonds with water and therefore they were drawn together).

4.1.1.2. Exocellular polymers (ECP) characteristics

Exocellular polymers (ECP) are compounds excreted by the bacteria. They have a form of numbers of protruding and mutually interlaced polysaccharidic or glycoproteinaceus fibers, forming around the cell or a group of cells a felt-like envelope called a glycocalyx (Costerton and Irvin, 1981). Glycocalyx forms the outermost surface of activated sludge. The content of the glycocalyx is different depending on the type of the organism. It comes from the metabolism and lysis of microorganisms (proteins, DNA, polisaccharides and lipids) as well as from the wastewater (Urbain et al. 1993).

ECP can be divided into two categories: polymers in the bulk liquid and polymers attached to the cell surface or incorporated in the flocs. Extracellular polymers consist of high molecular weight compounds (M_w > 10 000) such as polysaccharides, protein and nucleic acids. Matrix of extracellular polymers has been defined as those materials which can be removed from microorganisms without disrupting the cells and without which the microorganism is still viable. The ECP have different functions: conserving and concentrating the digestive enzymes, being a food reservoir, protecting against stress caused by protozoan, viruses, harmful ions and molecules and grouping microorganisms into an organized community. Measurements of the ECP yield in relation to time showed that, although the yield increased with extraction times of up to one hour, after this the yield was constant (Clarke and Forster, 1982). Polymer production has been found to be at a maximum during the endogenous phase of growth (Pavoni et al. 1972).

Hydrophobic/hydrophilic properties of activated sludge ECP have been investigated by Jorand et al. (1998). The authors found that 7% of ECP (expressed as DOC) is hydrophobic and consists of proteins and carbohydrates. Their research supported the hyphotesis of hydrophobic ECP participating in the organization of flocs.

Magara et al. (1976) have looked into a settling characteristics of an activated sludge from the physical and biochemical point of view. Corresponding to worsening of settling characteristics, changes of the physical properties such as the size, density and floc strength, and also the biochemical properties such as the ECP, PHB (poly - β- hydroxybutyrate) and electrophoretic mobility were observed. They stated that the settling characteristics of the activated sludge become poor with an increase of ECP or PHB. The limiting values of these indexes to maintain a good activated sludge were 5 mg/l of ECP and 10 mg/l of PHB. The authors stated that the ECP may be an adequate index for the settling characteristics of the activated sludge. ECP have the capability of sorbing significant quantities of trace heavy metals from solutions. Parsons et al. (1971) reported that ECP of *Zooglea ramigera* are the extracting agents that remove significant amount of dissolved copper, cobalt, iron, and nickel from aqueous solutions.

Eriksson and Alm (1991) studied the electrostatic interaction between bacterial surfaces, ECP and polyvalent metal ions adding different concentrations of EDTA (complexing agent) to activated sludge samples. The settling tests indicated that the polymers interact with the primary flocs by means of metal ions and that this takes place at the outer surface of the primary flocs.

Since bacterial surfaces and ECP provide negative adsorption sites, the role of divalent cations on the floc stability should be emphasized. Divalent cations such as Ca^{+2} and Mg^{+2} are known to be involved in the chemical structure of bacterial aggregates (Eriksson and Alm, 1991) because of their ability to bind to negatively charged chemical groups. The cations showed higher affinity for exocellular polymers than for the whole sludge. A difference in affinity between Ca and Mg was also reported by Forster and Dallas Newton (1980). Bruus et al. (1992) discovered that the extraction of Ca^{2+} from activated sludge flocs caused an increase of the number of small particles and decreased sludge dewaterability. Urbain et al. (1993) concluded that exocellular polymers from metabolism, cell lysis or wastewater are involved in the formation if a three dimensional matrix or gel where divalent cations Ca and Mg acts as bridging agents with probably specific affinities for each kind of ECP. A high amount of exocellular polymers was associated with poor settling of the sludge.

Higgins and Novak (1997) and Murthy and Novak (1998) have shown that the excess monovalent cations can cause a deterioration in floc structure.

4.1.1.3. Methods of ECP characterization

Sludge surface composition was analyzed using many different methods. The extraction of the polymeric material can cause lysis of the cells which would contaminate the sample with the intracellular polymers. Typical techniques include: shearing, centrifugation, sonication and heat extraction by boiling in an alkaline solution. Thermal treatment was found to be the most effective extraction method for activated sludges. Chemical extraction methods caused most cellular disruption and ultrasonication released only low concentration of EPC (Brown and Lester, 1980). Since there is not unified method for the extraction of exocellular polymers it is difficult to compare results from different studies.

4.1.1.4. ECP and sludge settleability

There is an abundant evidence that the settling properties are related to sludge chemical structure and nature, especially with regard to the surfaces of the sludge matrix and the component microbes. If different nutritional conditions can promote (or suppress) the proliferation of filamentous bacteria which in turn are claimed to cause bulking (Wagner, 1982b) it is also possible that nutrients and the molecular weight of the sludge ECP influence (or are influenced by) the settlement characteristics of activated sludge.

The most significant results seem to be reported by Wu et al. (1982). They showed that settling properties, surface charge, the floc ecology and the amounts of biopolymers in the sludge (both protein and carbohydrate) all depended on the balance of the nutrients in the substrate and the rate at which the substrate was fed to the reactor. Forster and Dallas-Newton (1980) have suggested that the settlement properties are related to nutritional conditions. They suggested that a logarithmic relationship existed between the settlement index and each individual nutrient and also a new factor, the specific nutrient product (SNP). This was defined as:

$$SNP = (\text{Flow rate})\,(BOD)\,(NH_3)\,(PO_4)\,(\text{Aeration Volume})^{-1}\,(MLSS)^{-1} \qquad (15)$$

In another paper Forster (1985a) looked into the binding of polyvalent metal ions (copper, zinc, nickel and chromium) exocellular polymers extracted from activated sludge. The experiments were conducted on the sludge from full-scale treatment plants with different settling characteristics. The binding properties of ECP to polyvalent metal ions were examined by gel filtration and equilibrium analysis. It was found that the number of binding sites for metal ions varied randomly with SSVI. Urbain et al. (1993) found a positive relationship between SVI and the amount of extracellular polymers i.e. high concentration of ECP resulted in decline of sludge settleability.

Andreadakis (1993) studying the carbohydrate content within the sludge flocs tried to correlate it with the sludge settleability. He found that the carbohydrate content varied between 6-18% but there was no correlation between the carbohydrate content and sludge settling properties. In view of his results it is possible that sludge settleability is more affected by the properties of the exocellular polymers then the amount of ECP .

Forster (1985a) examined the settling characteristics of activated sludge in relation to the variation of the main nutrients (carbon, nitrogen, phosphorus) using data from a number of full scale plants. The results tended to confirm previous works (Table 6) that there were certain combinations of nutrients that were more likely to produce poor settling influencing both the species present in the sludge and also sludge surfaces. However, the authors concluded that any further work to asses the precise nature of this influence was likely to be expensive and not necessarily productive. The authors also suggested that the SSVI increased with the increase of the molecular weight of the major fraction of activated sludge polymers (whose weight was<100 00). The molecular weights of the surface polymers extracted from activated sludge by heat were determined by a gel-filtration unit with an exclusion limit of 100 000.

Table 6. Existing relationships between sludge settlement and nutrients (Forster 1985)

Source	Relationship
Hattingh (1963)	SVI$\propto$ (BOD/N;BOD/P)
Forster (1968)	SVI$\propto$ (NH_3 -N/PO_4-P)
Forster and Dallas - Newton (1980)	SVI$\propto$ (BOD;NH_3-N;PO_4-P;MLSS)
Wagner (1982a)	SVI$\propto$ (P/MLSS)
Wagner (1982b)	SVI$\propto$ (P/N)
Clark and Forster (1983)	SVI$\propto$ (BOD;NH_3-N; PO_4-P;MLSS)
Wu et al. (1982)	SVI$\propto$ (COD/N; sludge loading rate)

4.2. Physical chracteristics of activated sludge flocs

Activated sludge flocs are an important research subject, as their physical state and properties influence the mass transport and the effectiveness of separation of wastewater from the activated sludge. The flocs are aggregates of suspended solids containing different groups of microorganisms resulting from physicochemical and biochemical flocculation. They are irregularly shaped, fragile and have a high water content. Particles in activated sludge vary from a single microorganism cell, the size of a few microns or less to a large number of cells up to several thousand microns in size.

Finstein and Heukelekian (1967) considered flocs as particles with diameters larger then 20μm. Javakeri and Dick (1969) defined a floc as a group of primarily particles and the liquid within it and an aggregate as a cluster of flocs and the liquid between them, in most other works the terms floc and aggregate are used interchangeably.

Activated sludge flocs have a very complex structure and there are various techniques available to describe their physico-chemical structure. Urbain et al. (1993) have made a summary of the most common methods to describe floc characteristics. The various factors normally studied include: amount of filamentous microorganisms, floc size, surface charge, amount of extracellular polymers, amount of divalent cations (e.g. Ca^{2+}, Mg^{2+}), floc strength, floc density and hydrophobicity. Barber and Veenstra (1986) found filament length to be single the most important factor among volatile fraction, bound water content, electrophoretic mobility, filament length and quantity and floc particle size and quantity in gravitational thickening.

So far approximately 13 physical characteristics of activated sludge flocs have been studied and reported in the literature. In this report definitions and description of only the most typical ones will be reviewed.

4.2.1. Floc size.

Because of the irregular shapes of the activated sludge flocs their sizes have been described in many different ways. Ganczarczyk (1969) and Sezgin et al. (1978) took the floc length or maximum dimensions as the criterion of the floc size. The size of the flocs have also been described by their perimeter, circumference or breadth (Mueller et al. 1967; Li and Ganczarczyk ,1985). The range of floc sizes of activated sludge as reported by various workers varies between 0.5 and 1000 µm (Knudson et al. 1982) but most of the flocs are smaller then 100 µm. Sezgin et al. (1980) found that in the lab-scale activated sludge units the mean floc size varied from day to day. Among data taken in a period of 87 days, in an air diffusion system, the mean floc sizes were in the range of 21 to 190 µm with an overall mean of 85 and a peak value floc size of 275 µm; for oxygen diffusion system and 84 days period the values were: 28 to 297 µm; 120 µm; and 820 µm, respectively. Li and Ganczarczyk (1988) reported 80% of the flocs smaller then 100 µm, a percentage which is probably higher since photographic technique they used had difficulties with detecting flocs <25 µm.

Sizing results may be altered by the improper sampling procedure changing (through breakup or compression) the structure of the flocs. Sizing techniques vary from study to study and various methods for particle size measurements are available. Most of the methods assume one or two principal axes and sometimes a sphere-like shape, and because of three dimensional structure of the floc realistic sizing is still difficult. The first method accepted for measuring the floc sizes was a microscope associated with different micrometer scales for calibration. This method allows to measure the floc size by direct observation; flocs are usually sized and counted in a series of size intervals. Flocs of >800 μm could be sized individually (Sezgin et al. 1978).

The use of photography for floc size measurements was reported in 1955 by Schopfer (as cited by Mueller et al. 1967). Since then many researchers (Magara et al. 1976; Tambo and Watanabe 1979; Li and Ganczarczyk 1985) applied this technique for measuring both activated sludge and inorganic floc sizes. Photographic technique allows to measure the floc size without directly touching the samples, which makes this technique very attractive. Li and Ganczarczyk (1987) introduced a multi-exposure photographic (stroboscopic) method for measuring of settling velocity and the size of activated sludge flocs. The smallest investigated flocs were about 50 μm. The authors observed that the settling velocity of the flocs was linear with the floc size (assuming a cross-section diameter as a criterion). The porosity of activated sludge flocs increased at a higher rate when floc sizes were smaller then 200 μm (longest dimension) as compared with that of the flocs larger than 200 μm.

Other methods include image analyzing system (Zahid and Ganczarczyk, 1990; Li and Ganczarczyk, 1991), Coulter counter (Li and Ganczarczyk, 1991; Andreadakis, 1993) and laser beam diffraction (Jorand et al. 1998). The different measurement techniques presented by Kavanaugh et al. (1980) are summarized in Table 7 (after Willén, 1995). Similar excellent review of the different measurements methods and techniques was presented by Li and Ganczarczyk (1985).

Table 7. Summary of different size measurement techniques

Measuring principle	Equivalent of size measured	Size limits	R

		minimum	
1.Electron microscopy	statistical length	0.001	50-200
2.Optical microscopy	statistical length	0.3	40
3. Coulter counter (electrical sensing zone method	volume diameter	1	20
4. Light scattering laser method	cross section diameter	1	10-50
5. Light obscuration	cross section diameter	1	50-60

R - ratio of maximum to minimum size for single sensing element, or single magnification

The other methods found in the literature include measurement of turbidity and its further calibration by the average floc size which was obtained photomicrographically (Riech and Vold, 1959) and various filtration techniques. However, the latter ones have generally had poor results due to clogging problems.

The particle size and specially their size distribution are the result of dynamic equilibrium state between formation, transformation and breakage of microbiological aggregates. In their later works Li and Ganczarczyk (1991) looked into the size distribution of the activated sludge flocs from different wastewater treatment plants. Flocs smaller then 10 μm were measured by a Coulter counter and larger ones by an image analysis system. Flocs larger than 10 μm fitted best the log-normal model while flocs larger then 50 μm were the major source of surface area, volume and mass although they were fewer then the smaller ones.

Snidaro et al. (1998) studied the size distribution of sonicated activated sludge flocs. They concluded that the model of floc structure is made up of three basic elements. Investigation by confocal scanning laser and transmission laser microscopy showed that they include:

- 2.5 μm units - primarily living bacteria cells;
- 13 μm units - aggregates of the individual particles that are encapsulated in a polymer matrix to form microcolonies;

- 125 μm units - aggregates of microcolonies. A gel- like matrix of extropolymers provided the cohesion of these units; also channels and pores exist throughout this level of structure and flow has been observed within these channels.

To retain the specific size distribution of the flocs and the geometric properties of the floc some kind of physical stabilization of the microbial aggregates is required. An improved technique for the physical stabilization of microbial aggregates, by embedding them in solidified agar was recommended by Ganczarczyk et al. (1992) for light microscopy studies such as the measurement of size and the evaluation of morphological parameters.

Another study in this area was conducted by Andreadakis (1993). He tried to evaluate in 7 batch-scale experiments physical and chemical characteristics of activated sludge including: floc size, density, specific surface, carbohydrate content, dehydrogenase activity and settleability. The experimental units were operated at different values of a sludge age ranging from 1.1 to 17.4 days. The typical floc sizes (more then 85% of flocs) were found to be in the range 10 - 70 μm with the median values between 35-45 μm. The author observed a strong correlation between floc density and size.

Further work of Ganczarczyk and his co-workers (Li and Ganczarczyk, 1992) was directed toward proving that the settling velocity of microbiological aggregates is a function of their geometrical parameters as well as their density and porosity. They recognized that permeability is related to porosity and developed the relationship to describe the force balance on porous aggregates of spherical and filamentous particles.

The chaotic nature of flocculation and breakup of the flocs caused that fractal geometry was introduced into description of particle aggregates in wastewater treatment process (Li and Ganczarczyk, 1989). A fractal structure is considered a form where the density/porosity is a function of a characteristic aggregate dimension which is used to define the aggregate geometry. Floc aggregates are now recognized as fractal objects with the main parameter called the fractal dimension D. This parameter represents the distribution of the mass in a fractal object and can be calculated as a slope of the line obtained after plotting the mass M of a fractal against aggregate size R (diameter).

$$M(R) \text{ is proportional to } R^D \tag{16}$$

The fractal dimension can take any positive value less then 3; fractal dimensions of biological floc aggregates in the research reported by Li and Ganczarczyk ranged between 1.4-2.8. The lower the fractal dimensions, the more open is the aggregate structure. The fractal theory can be used for analysis of geometric characteristics of the aggregates, substrate transfer in biological flocs and the flocculation process. The activated sludge structure was analyzed using the fractal dimension procedure by Sniadro et al. (1998).

Floc size can be affected by the following factors: sampling procedure, microorganisms present, agitation, dissolved oxygen and organic loading, basin scale, particle concentration, sludge age and nature of substrate (Li and Ganczarczyk, 1985).

4.2.2. Density

The density of the activated sludge influences both the diffusion of the substrates into the floc and the sludge settling characteristics. The particle density and therefore the biofilm density can be calculated on the basis of particle settling velocity and size measurements. The values of activated sludge density vary depending on the way the density of the solids in the suspension is considered (density of aggregate particles and water between them, density of single flocs and water between them, density of the flocs only). Mueller et al. (1967) considered that the sludge density should be the one after removal the associated water and calculated that the value of mass sludge density was 1.09 g/ml. Li and Ganczarczyk (1987) measuring the size and settling velocity calculated the density of the activated sludge. A typical value for dried sludge was 1.4 g/ml.

Changes in floc density are usually small. A concept of effective density, i.e., the buoyant density was defined by Magara et al. (1976) and Tambo and Watanabe (1979) as

$$\rho_e = \rho_p - \rho_m \tag{17}$$

where: ρ_e -effective density of flocs, g/ml; ρ_p - density of particles, g/ml; ρ_m - density of water, g/ml has been used in most floc density studies (Li and Ganczarczyk, 1985). The results of almost all studies of the size-density relationship for inorganic and biological flocs showed that the floc effective density would decrease as the floc size increased.

Moudgil et al. (1989) looked into the role of the floc specific size , density and shear strength in the flocculation of chemical suspension process trying to establish techniques that would yield flocs of desired properties for a given polymer/particle system. Within the range of polymer dosages examined the amount of flocculant had no significant effect on floc density.

Dammel and Schroeder (1991) developed a density gradient centrifugation method to measure the activated sludge floc density. The measured particle densities, for the seven activated sludge treatment plants observed in this study, varied from 1.02-1.06 g/ml. Measured density increased linearly with ionic strength of the liquid medium. A clear relationship between solids density and mean cells residence time was not found. Andreadakis (1993) in his work calculated the bulk density of the floc as ranging from 1.015-1.034 g/cm^3; the value was in agreement with values found for moderately and low loaded activated sludges.

Lee et al. (1996) have estimated the feasibility of using free-settling test for estimating activated sludge floc density. The information obtained in this test is the diameter and the terminal velocity for a single floc. Assuming that the floc in moving steadily in the medium, the force balance equation is used for estimating floc density (and also floc porosity). The terminal velocity and diameter data for a total of 1385 flocs original or treated by a freeze-thaw treatment were found experimentally. The primary particle density was estimated as ranging from 1450 kg/m^3 to about 1045 kg/m^3 for original sludges or 1070 kg/m^3 for frozen sludges.

Porosity is also considered as one of the parameters determining activated sludge floc characteristics by describing those spaces in flocs occupied by water rather then by solids. Activated sludge flocs are highly porous in structure because of the high water content. Tambo and Hozumi (1979) conducted the analysis of aluminum floc-binding force and from floc material balance calculated the porosity as:

$$\rho_w V_e + \rho_0 \ (1- e)V = \rho_f \ V \tag{18}$$

where: V - volume of flocs or particles, mm^3/floc; p_f, p_w, p_0 - density of the floc, water and primary particle respectively, g/cm^3; e - porosity; they assumed that the floc consisted of two parts: primary particles and water so the porosity could be defined using the following formula:

$$e = 1 - \frac{a}{(d/1)^{K_p}(r_p - r_m)} \tag{19}$$

where: K_p and a - coefficients with units g/cm and dimensionless, respectively; d/1 - the relative floc diameter. The similar approach was followed by Li and Ganczarczyk (1987). They reported that the porosity of activated sludge flocs increased at two different rates in a range from 43 to 99.6%. For smaller flocs (less than about 200µm) the porosity increasing rate of was much higher than that of the larger flocs.

Porosities of the flocs presented by Andreadakis (1993) in his research varied from 85-97%. Non- filamentous sludge settleability, expressed in terms of SVI was related to floc size and density. A strong correlation between SVI and floc specific surface area was also observed. Growth and competition may also contribute to the development of a porosity gradient within the microcolony. Scuras et al. (1998) presented a model to predict microcolony porosity and diffusivity and their impact on growth and competition between the microorganisms. They found out that there is a rapid progression toward minimum porosity with relatively slow expansion of the microcolony surface.

4.2.3. Floc strength

There are few reports on the floc strength of an activated sludge because there is no effective way to determine it. Floc strength cannot be observed directly and various methods were developed to estimate its magnitude. It can be measured directly through determination of the extent of floc damage under certain conditions and effectively used for floc strength studies by comparison assuming that the floc samples are damaged at the same destroying conditions. The indirect methods determines the floc strength by use of some coefficients or process parameters based on measurements of certain variables of an activated sludge process.

On the other hand Tambo and Watanabe (1979) recommended a new index of floc strength as a result of research on an inorganic floc such as aluminum hydroxide. The strength of the aluminum hydroxide floc is given as that relative to a certain standard floc whose properties are stable. They suggested that the floc strength could be derived from the force balances on the floc assuming that breakdown of the floc occurs when the sheering stress which is caused by a turbulent fluctuation exceeds the floc binding force. In their later work Tambo and Hozumi (1979) developed theoretical and experimental methods to evaluate a maximum floc diameter attainable under an agitation condition and to express a function of floc strength as the binding force per unit area. The relative floc binding strength of several flocs with respect to a clay-aluminum floc was calculated.

Magara et al. (1976) followed their assumptions and attempted to find the strength of an activated sludge floc. Their results reveal that the floc strength of the activated sludge is 3-6 times higher than that of an aluminum floc. An activated sludge floc also keeps its size more than a chemical floc. Moudgil et al. (1989) evaluated the strength of kaolinite, dolomite and Al_2O_3 flocs with various polymers. They discovered that the optimum binding strength was found to be independent of the molecular weight of the polymer; the dosage however was determined to be 2.5 times lower for the higher molecular weight polyacrylamide.
On the basis of theoretical considerations Parker et al. (1971) developed a differential equation describing the net rate of change of the number of primary particles (n) with regard to time in a batch reactor:

$$dn/dt = k_B X G^m - k_A X n G \qquad (20)$$

where: n - primary particle number concentration, 1/l; t - time, s; X - MLSS concentration, mg/l; G - root- mean- square velocity of shear gradient, 1/s; k_a- floc aggregation rate coefficient, l/mg; k_b- floc breakup rate coefficient, s^{m-1}/mg; m - floc break-up rate exponent, dimensionless. Further work in this topic were conducted by

Wahlberg et al. (1994) who developed this equation and reported a survey of the flocculation parameters measured at 21 treatment plants.

4.2.4. Surface charge

Surface charge most frequently develops through a preferential adsorption or ionization. In the case of microorganisms surface charge is acquired through the ionization of carboxyl and amino groups. In the wastewater environment and its almost neutral pH range the microorganisms have a negative charge. Regardless of how it is developed this stability must be overcome if these particles are to be aggregated into larger particles. The surface charge is balanced by the counter ions (ions of the opposite charge) that are being attached to its surface by electrostatic and van der Waals forces creating the electrical double layer.

The potential at the surface of the cloud of the counter ions is called the zeta potential and its value determines the ability of the particles to form aggregates. The zeta potential is measured as electrophoretic mobility of the particles (velocity of particles in the electric field). The negative surface charge can be estimated by the adsorption of cationic molecules such as ruthenium red (RR^+) on the surface of the flocs (Figueroa and Silverstein, 1987; Urbain et al. 1993; Piirtola et al. 1998; Löwen and Piirtola, 1998)).

Pavoni et al. (1972) showed that the surface charge reduction is not the prime mechanism in bioflocculation because polymers are able to bridge the cells either electrostatically or physically. Barber and Venestra (1986) could not find any relationship between electrophoretic mobility of sludge particles and SVI of activated sludge. Forster (1968) found a linear relationship between electrophoretic mobility and SVI while Magara et al. (1976) reported an inverse relationship between SVI and electrophoretic mobility.

Out of many physical properties of activated sludge only floc size number and length of the filaments, floc settling velocity and density have been well defined and can be measured directly. The image analysis methods probably give the highest degree of accuracy for floc size and filament content. There are many well-developed methods for calculation of floc water content, porosity and floc strength. However, it is difficult to measure directly these parameters for individual flocs.

Many activated sludge process variables and operating conditions cause unexplained changes in physical characteristics of the activated sludge flocs. The correlation between those phenomena should be further researched and discussed.

4.3. Microscopical methods

The activated sludge flocs are the most important element of the wastewater treatment process. Apart from checking the course of treatment process through chemical and physical analysis of influent and effluent, the standard investigative procedure should include also the actual monitoring of the sludge quality. Moreover, regular microscopic examination of sludge helps to check the treatment process accurately and detects a deterioration of the sludge quality before problems occur on a large scale.

Microscopic sludge examination was originally developed so that changes in the amount and the type of filamentous organisms with time could be recognized. However, it has been shown that the value of these analyses can be increased considerably, when the investigations are not restricted to these organisms. Many other changes in the floc composition can be noticed simultaneously, like: shape, size and structure of the flocs, its composition, filamentous microorganisms and ”higher organisms”, dispersed growth of bacteria etc. (Eikelboom, 1982).

However, estimation of sludge parameters related to the bulking of sludge still remains as the main element of microscopic examination of sludge. Eikelboom (1977,1981) has made a great breakthrough in the investigation of activated sludges filamentous bulking and foaming and his method was further modified by Jenkins et al. (1984). The techniques are based on phase contrast microscopic observations of morphology, relationship to other organisms present and staining characteristics. The filamentous organisms are classified into types according to the following features: cell shape dimensions, presence of sheath, filament morphology staining characteristics and presence or absence of polyphosphate and PHB granules. With few exceptions the taxonomic position of most Eikelboom's types is still unknown and the types are characterized by numbers (e.g., Type 0041 or Type 021N).

Eikelboom (1982) divided also activated sludge into so-called "categories " (five categories) on the basis of the extent of filamentous growth. In ascertaining the category of a certain sludge the microscopic view is compared with the series of reference images in the form of photographs of the different categories. This method of standardization has been employed by many water authorities in the Netherlands.

Apart from some characteristic morphological features the identification to types is based on reactions to Gram and Neisser staining (Eikelboom and van Buijsen, 1981; Jenkins et al. 1993). However, the morphology and staining reactions may vary significantly depending upon cultivation conditions and wastewater composition causing some uncertainty in the results of microscopic identification. A typical example of this uncertainty is the most problematic filamentous microorganisms *Microthrix parvicella* when the scientists studying this filament cannot agree at present that all the laboratories all over the world work with the same microorganism (Foot et al. (1992) examined the morphological characteristic of *Microthrix parvicella* grown under conditions which approximate to those that can occur in domestic sewage treatment systems. The results showed that the filament display both variable morphology and variable Gram stain reactions when fed on selective substrate. Long Gram-positive filaments broke down within a few days into short Gram- negative filaments and single cells with Neisser-positive granules. Changes in morphology of *M. parvicella* were linked to specific substrate loading rate while the Gram stain variability was related to the storage of lipids and protein synthesis. Such variable taxonomy causes separation problems in laboratory and full scale activated sludge systems.

A comprehensive review of different techniques and media used for isolation of various filamentous bacteria and morphological and cytological characteristics of the isolates was presented by Ziegler et al. (1990).

Soddel et al. (1993) in their paper looked for one more accurate and applicable in practice method and suggested gene probes for *in situ* identification and quantification of filamentous organisms. Blackall (1993) and Wagner et al. (1993) proved that such an identification is feasible for activated sludge filamentous microorganisms. The molecular identification is based on specific sequences of ribosome 16S and 23S rRNA/DNA which can be detected by hybridization with fluoroscently labeled oligonucleotides. With this technique Blackall (1993) showed that the strains of foam-forming microorganisms *Nocardia amareae* and *Nocardia pinensis* most probably belong to the genus *Nocardia* although there were some doubts in the literature about this allocation.

Wagner et al. (1993) succeeded to identify *Spherotilus natans*, *Haliscomenobacter hydrossis* and *Thiothrix nivea* by using *in situ* hybridization technique with 16S rRNA target probes. Hernandez et al. (1993) developed an immunofluorescent method to estimate the quantity and viability of Nocardia filaments in activated sludge and anaerobically digested sludge on both a mass and volume basis. Using immunofluoreprobes specific to *Nocardia* Hernandez et al. (1994) quantified the mass of *Nocardia* in activated sludge and anaerobic digested samples. Their study showed that *Nocardia* biomass accounted for 10-28 % of VSS in a full- scale activated sludge system and for 8-19 % in anaerobic digested sludge. Quantification using Gram-staining and filament counting for the same activated sludge sample resulted in a significantly lower *Nocardia* representation (0- 2 % of the total VSS).

Apart from filamentous organisms identification method there was also a need to develop some measures to estimate their quantity and length within the activated sludge floc so that the numerical value can be compared with SVI or SSVI. Many workers have used systems of subjective assessment such as the ”+ to +++” scale of Rensink (1974), or the arbitrary 10-point scale of Forster and Dallas-Newton (1980). Chudoba et al. (1973) estimated the relative proportion of the microscope field occupied by filamentous and zoogleal bacteria. Basically, filament counting and sizing make use of a light microscope associated with some calibrating meters (Finstein and Heukelekian, 1967; Sezgin et al. 1978; Pipes, 1979).

Further these method were modernized by Walker (1982) and Nowak et al. (1986) by using an improved Neubauer hemacytometers for filament counting. A grid superimposed on to the image of the activated sludge under the microscope allows to calculate the length of filament per unit weight of dry solids (if the MLSS is determined in the same sample). There is a drawback to these filament -sizing and counting methods. As mentioned in the report by Sladka and Zahradka (1971) the actual filament content was much larger then estimated since part of the filament located on the upper and lower sides of the floc could not be seen.

4.4. Sludge Indices

In wastewater treatment plants sludge thickening occurs in the lower portions of clarifiers and in separate thickening tanks. Due to wide variation in sludge settling properties both final clarifiers and thickeners should, if possible, be designed on the basis of pilot plant data. Continuous -flow pilot units are expensive and difficult to operate so design criteria for the process are mostly based on batch thickening tests.

A number of parameters have been developed to obtain a quantitative measure of the settleability of activated sludges. All of these tests are based on one of two basic approaches. The first approach uses the volume of the sludge occupied after a fixed period of settlement. In this approach laboratory tests are conducted by allowing a sludge to thicken in a 1 liter graduated cylinder, with or without stirring. The position of the sludge water interface is recorded. The second approach uses the subsidence velocity of the solid/ liquid interface of the sludge at its initial concentration calculated from the straight -line portion of the resulting curve.
Different measurement parameters of sludge settleability are characterized in the following chapters.

4.4.1. Sludge Volume Index (SVI)

The traditional sludge volume index (Mohlman sludge index) is the mainly performed test. The test is easily performed and has a widespread use in routine process control. The test apparatus is a 1 liter sedimentation vessel. Normally the test is performed without stirring although stirring is recommended by the Standard Methods (1981). Calculation of SVI is done by the equation:

$$SVI\ (ml/g) = SV/SS \tag{21}$$

where: SV - the sludge volume after 30 minutes sedimentation in a 1 liter sedimentation vessel, ml/l;SS - initial suspended solids concentration, g/l.

The shortcomings of SVI have been discussed by Dick and Vesilind (1969). Their data show that SVI does not relate to such parameters as sludge yield strength, plastic viscosity, and an initial settling velocity. The variation of SVI values with the suspended solids concentration and the sludge volume makes it difficult to compare the SVI values for instance between different plants. Two sludges with the same SVI may have different sedimentation and dewatering properties. At high solids concentration the settled volume of the sludge, as measured in a unstirred 1 litre cylinder, increases at approximately the same rate as the increase in MLSS thus suppressing the SVI. For well settling sludges the concentration above which the SVI becomes strongly influenced by the concentration is relatively high (about 6 g/L), while for poorly settling sludges the critical concentration can be lower then 2 g/l. The value of the SVI depends also on whether the sample is stirred or not during the test (the procedure for conducting the SVI outlined in *Standard Methods* was modified in 1980 to include slow stirring , between 1-2 r/min) since stirring cuts the bridges between the sludge flocs and gives therefore a lower SVI.

Pipes (1979) reported that high SVI is usually the result of filamentous microorganisms presence in the sludge and that higher SVI usually produces a lower effluent TSS. The relationship between SVI and organic loading has been investigated for instance by Ford and Eckenfelder (1967), Ganczarczyk (1970), Rensink (1974), Pipes (1979). They found low SVI within the intermediate range of the loading 0.2 - 0.4 g /g d.

The sedimentation vessel diameter also affects the settling velocity. Vesilind (1968) has found that a suspension with a low concentration of SS may settle more rapidly in cylinders of small diameter than in a large cylinder because of liquid streaming up the walls, whereas a suspension with a high concentration of suspended solids may settle more slowly in a small cylinder than in a large one because of bridging across the walls and partial support of the solids from below. Normally the diameter of the test is not specified. Dick and Ewing (1967) have shown that the initial height of the suspension is important and that the settling velocity may be retarded at shallow depths (1 liter cylinder) by partial support through the solids from the bottom of the vessel.

Recently Bye and Dold (1998) presented a simple model for predicting behavior in a column settling test that can be used to generate an SVI-solids concentration relationship for a given solids sample. The model can be used as a basis for evaluating the effects of different variables (test column height, solids compactability and zone settling characteristics) on SVI-results. Completing the models the authors agreed that SVI shows a dependence on the solids concentration and is not a good measurement for monitoring sludge bulking. In spite of these weaknesses the wastewater treatment plant operators keep using the SVI successfully on a day-to-day basis. Sekine et al. (1989) have developed a SVI-meter and data analyzer for use in full scale plants on line-measurements

4.4.2. Stirred Specific Volume Index (SSVI)

In an attempt to overcome the deficiencies in the SVI the Water Research Centre (WRC) has developed a Stirred Specific Volume Index (SSVI) that tries to reproduce the non-ideal situation in settling tanks whereas the SVI is measured under complete quiescence (White,1975). For most of the sludge investigated the SSVI was independent of the initial suspended solids concentration. However, for some sludges with poor sedimentation properties this was not the case. Therefore it was suggested that a standard concentration of 3.5 g/l should be used for reporting the SSVI data, denoting it as $SSVI_{3.5}$.

White suggested that the test apparatus was 3.5 l sedimentation vessel (as opposed to the *Standard Methods* suggestion of a 1-l settling column) of 10 cm external diameter, fitted with a low-speed stirrer, which rotates at 1 rpm. The standard SSVI is obtained by doing a number of SSVI tests over a range of sludge concentrations from 2 to 6 g/l and interpolating the value at 3.5 g/l. The SSVI (ml/g) is calculated as:

$$SSVI_{3.5} = SSV/SS, \qquad (22)$$

where: SSV- sludge volume with stirring, ml; SS- concentration of suspended solids (g). White even developed nomographs for using the information on SSVI to modeling on the flux theory approach; good settling occurs with $SSVI_{3.5}$ values up to

120 ml/g, poor settling or bulking in excess of 200 ml/g. Subsequent research has indicated that the SSVI test is relatively insensitive to settling column dimensions providing that it is not smaller than White's specifications, i.e. a depth to diameter ratio of 5:1 to 6:1 (White, 1976; Rachwal et al. 1982; Pitman, 1984).

The SSVI test became more widely used for a normal operation management as the SVI has shown to have serious limitations and some marked changes in settleability which have been reflected by the SSVI have not been detected by SVI (Rachwal et al. 1982). The authors conducted extensive sludge settling tests in which they measured SSVI and the flux theory constants V_0 and -n. Ekama and Marais (1986) based on their research showed that both the SSVI and flux constants are relatively insensitive to column dimensions, probably owing to the gentle stirring.

The SSVI has been widely adopted as the principal settleability measurement in England and is used for the design and control of secondary clarifiers. There are several reasons of its popularity (Ekama and Marais, 1986):

- Reproducible results for the relationship between SSVI and solids concentrations up to 10 g/l for good settling sludges and 7 g/l for poor settling sludges (though White's (1975) observation suggested that the correlation between SSVI and settling rate is poor for concentrations less then 3.5 g/l).
- The 30- minute settled volume fraction may be as high as 0.7-0.8 without adversely affecting the test results
- The relationship between the 30- minute settled volume fraction and solids concentration is close to linear.

4.4.3. Diluted Sludge Volume Index (DSVI)

Stobbe (1969) and Lee et al. (1983) investigated the dependence of the SVI value on the suspended solids concentration and showed that the test result from a test performed in a 1 liter cylinder is approximately independent of the sludge concentration if the sludge volume after 30 minutes is less than 250 ml/l. Based on the results Stobbe suggested the use of the DSVI in which test the sludge is diluted with secondary effluent in such a way that the sludge volume after 30 minutes is less than 250 ml/l i.e. 200 ml/l (20%). Such volume was chosen arbitrarily as a value that would ensure that ”constant” SVI conditions was reached for most samples. The test apparatus is a 1 liter sedimentation vessel and the index calculation is:

$$DSVI = SV/SS \tag{23}$$

for SV < 250 ml/l.

Its insensitivity to the sludge concentration makes it possible to compare sludge settleability in different activated sludge plants and incorporate DSVI into secondary clarifier design procedures such as ATV or STOWA (see chapter 1.7). Also Koopman and Cadee (1983) developed a procedure that was a derivative of the flux theory, and linked, with the aid of empirical relationships between the DSVI and the flux theory coefficients V_0 and -n-, the DSVI to the flux theory predicted maximum solids loading rate. Their investigations confirmed that the DSVI and SSVI procedures produce reasonably comparable results.

According to the study conducted by Bye and Dold (1998) the DSVI ”forces” the SVI test conditions into the region at which SVI is independent of solids concentration. However in this region the observed SVI bears no relation to the settleability of the test sample. Rather the DSVI only provides a measure of solids compactability. Two samples of with very different zone settling parameters (V_0 and n) will have the same DSVI, however a correlation based on DSVI will predict the same V_0 and n. As a parameter reflecting the sludge compactability DSVI should provide a good information on sludge bulking characteristics.

4.4.5. Sludge Quality Index (SQI)

In order to diminish the influence of the initial concentration on the SVI, Fitch and Kos (1976) proposed modification in the calculation of the sludge index called the Sludge Quality Index (SQI). The test apparatus is a 1 liter sedimentation vessel and the calculation of SQI (ml/g) is:

$$SQI = SV/SS \tag{24}$$

for SV < 300 ml/l

$$\text{and } SQI = (200 + SV/3)/SS \tag{25}$$

for 300 ml/l <SV < 800 ml/l.

The test is performed without stirring. Different domains in sludge index determination according to SS is a way to overcome the difficulty of adjusting initial concentration.

The recalculation of the old data on SVI to the values of SQI is possible if the sludge volume after 30 minutes of sedimentation (SV) and the initial sludge concentration (SS) in a 1 liter sedimentation vessel without stirring are known. The SQI value is more reliable than the SVI value.

4.4.6. Modified Sludge Index (I_{VF})

Hultman and Molina (1987) suggested a new sludge index which takes into account the influence of the Volatile Fraction (VF) on the sludge sedimentation properties. The modified sludge index I_{VF} is compensated for the VF i.e. it is independent of the VF.
A new sludge index I_{VF} (ml/g) was defined as:

$$I_{VF} = (1 - VF)\ SQI/VF \qquad (26)$$

The purpose of this index is to get an index that is independent of the volatile fraction of the sludge and a function of the sludge structure only, such as the filament length per g MLSS. The calculation of the index has shown a good agreement with the literature data.

4.4.7. Bulking Index (BI)

Watanabe et al. (1990) have proposed the bulking index BI which might predict conditions for sludge bulking earlier than the SVI. BI decreases before SVI increases or sludge becomes bulking.
The calculations of BI (μm^2 mg/m ml) is:

$$BI = R^2/(FL / V) \qquad (27)$$

where R is floc size (µm), FL is the filament length (m/mg MLSS) and V is floc volume (ml). Values needed for bulking index calculation are quite complicated to measure.

4.4.8. Relationships between different sludge indices

Ekama and Marais (1986) reported studies of the correlation between the SSVI and DSVI and concluded that the SSVI value was approximately 2/3 of the DSVI value i.e. SSVI = 0.67 DSVI. This relationship was further revised by Ekama et al. (1997) and the authors concluded that a great caution should be exercised in using the relationship between SSVI and DSVI; statistically one is no more reliable than the other.

According to Randall et al. (1992) a relationship DSVI = 0.60 SVI is true when SVI is in the range from 160 to 220 ml/g. Hultman and Molina (1987) have shown that the SQI is approximately equal to the DSVI value. This has also been confirmed by Hultman et al. (1991). According to Hultman et al. (1991) the SVI values are considerably higher than the SQI and hence also higher than the DSVI values. In their studies with the sludge from Henriksdal WWTP the SSVI value was approximately 80% of the SQI value.

Based on the literature and experimental studies of Hultman et al. (1991) it is shown that the diluted sludge volume index (DSVI), the sludge quality index (SQI), and the stirred specific volume index (SSVI) are suitable to predict the sedimentation performance. Randall et al. (1992) recommended the use of DSVI as a settling measurement parameter. The traditional sludge volume index (SVI) is not as useful and should therefore be replaced.

The relationships between different sludge indices are gathered in the following table

Relationship	Reference
SQI = DSVI	Hultman and Molina (1987), Hultman et al. (1991)
SVI> SQI (for SV>300 ml/l)	Hultman et al. (1991)
SSVI = 0.8 SQI	Hultman et al. (1991)
SSVI = 0.67 DSVI	Ekama and Marais (1986)

DSVI = 0.60 SVI	Randall et al. (1992)

There have also been many attempts to find a possible relationship between filament content and size and sludge settling characteristics. Chudoba et al. (1973) used a microscopic technique that gave the relative proportion of filamentous and flocculated organisms in the sludge and found that the SVI was proportional to the proportion of the filamentous organisms. However, the filamentous bulking theory was formulated by Setzgin et al. (1978) and confirmed by Palm et al. (1980). They have measured the total length of filaments in sludge samples and shown that reflocculation time, compact volume of sludge and SVI increased proportionally to the total filament length when the filament content was over 10 m/ml; they also found that activated sludge bulking occurred when the filament concentration increased beyond 10 m/ml. According to the filamentous bulking theory formulated by Sezgin et al. (1978) sludges had poor settling characteristics as filament length extended beyond 10 m/ml.

First filament index system was introduced by Eikelboom and van Buisen (1981). Also Green (1982) tried to predict bulking of activated sludge by the development of a method to measure filamentous organisms. A MOP-VIDEOPLAN image analyzer was used to measure the length of the filaments recorded by photographs. The filament index (FI) was developed and correlated with sludge settleability data expressed as SSVI and SVI. The results were rather inconclusive and varied for different treatment plants.

Lee et al. (1983) investigated the relationship between filament levels and several quantitative indexes of sludge settleability (SVI, DSVI, SSVI at standard concentrations of 1.5, 2.5. and 3.5 g/l) for the purpose of the routine application of the filament content to the activated sludge process. The study used total extended filament length i.e. the content of filamentous organisms in the activated sludge expressed as a measured length per unit mass (TEFL km/g), as introduced by Sezgin et al. (1978). The results indicated that DSVI showed the best correlation with filament content and therefore it was concluded that DSVI was the best index of sludge settleability. The authors proposed a relationship between DSVI and the filament length as

$$DSVI = 56\,(TEFL)^{0.343} \quad (28)$$

where: TEFL - filament length km/g and concluded that a bulking sludge has a DSVI above 150 ml/g.

Also Kristensen et al. (1994) during their investigations of Danish nutrient removal plants tried to estimate the sludge settling properties through determination of SVI, DSVI, filament index and dominating filamentous organisms. They found a significant scatter when comparing filament index to SVI and to DSVI. It would indicate that other parameters such as floc structure, floc density etc. may be of significance for the settling properties of the sludge.

4.5. Sedimentation curves.

Attempts have been made to model the sludge sedimentation characteristics instead of using sludge indices determined by different sedimentation tests. The concept of using batch - settling data to design continuous thickeners has been used for many years, and while some theoretical development has been directed toward explaining the batch-thickening curve, there is still disagreement as to what is actually taking place during batch thickening.
Zone Settling Velocity (ZSV) of activated sludge is one of the experimental methods used for describing sludge settling, instead of SVI. ZSV is obtained from a solid/liquid interface depth-time plot and is given by the slope of the straight line part of the interface height versus time curve. The ZSV decreases as the solids concentration X increases. By conducting a number of stirred settling tests at different concentrations ranging between 1 and 12 g MLSS /l (at least 6) the ZSV at different X values is obtained. ZSV is well known but not as much used in daily monitoring of treatment plants as the traditional SVI. Measuring ZSV is laborious and probably that is why it has not become a standard method for determining sludge settling properties. The main constrain of ZSV is that it ignores the end of the settling curve and interprets sludge settling as a constant. The most important use of ZSV is probably in the clarifier models, specially in flux theory.

Ideal sludge settling conditions was first presented by Kynch (1952). His zone settling curve is composed of two stages: a constant - rate period, when the interface height versus time curve is a linear function, and a compaction period when the settling curve is curved (corresponding settling velocity is a variable and is generally less that the zone settling velocity). According to Kynch idealized concept of batch thickening, each concentration propagates at a characteristic upward velocity and eventually intercepts the interface. At the time of the interception, the interface assumes a settling velocity characteristic of the propagated concentration. When the maximum concentration reaches the interface, no further settling is possible. The interface velocity is constant until the first propagating layer reaches the interface at which time the velocity begins a steady decline to zero.

Numerous investigators have, however found that the Kynch ideal sludge is seldom found in practice and that the batch-thickening curve for most sludges is more complicated. Fitch (1975) presented the interface movement of a less idealized sludge, which is thought to be a more realistic representation of batch thickening. Initially there is an induction period that is followed by the linear settling portion, which is supposed to be characteristic of the initial solids concentration in the column. The first decreasing rate period is explained with Kynch theory as the propagation of higher concentration layers from the bottom to the interface. The second decreasing rate period begins as the compression region intersects with the interface.

The first significant difference between the two curves is the presence of an induction period, during which the velocity increases to some constant value. Coe and Clevenger (1916) among others have attributed the presence of this induction period to the formation of the solids structure most appropriate for settling.

The second major difference between ideal and more complex batch-settling curves is the second falling rate region. Since this is a compression region it can not be explained with the Kynch theory because that theory addresses only free zone settling and compaction. Dixon (1978) introduced another possible explanation for part of the transition from the first to the second decreasing rate period in the concept of dynamic (or transient) compression resistance. In a compressible sludge, dynamic compression resistance can be viewed as the flow resistance of the liquid being squeezed between the two particles.

Fitch (1975) used basic hypothesis about compression stress and plastic deformation presented by Michaels and Bolger (1962) to derive a relationship for the thickeners of the sludge beds in continuous thickeners. He felt that if thickening has a compression phase a substantial squeeze should have been detectable, which was not a case. Kos (1977) verified that compressive stress (squeeze) exerted in continuous thickeners were indeed small. But he also found that only small stress was needed to deform the flocs, so compression can make contribution to increasing solids concentration. Fitch (1983) later generalized the Kynch analysis and demonstrated that the characteristic lines (lines of constant solids concentration) must emanate tangentially from the sediment surface after the transition point.

Vesilind and Jones (1990) in their study further examined the interfacial settling behavior of pure chemical sludges at concentrations varying from 92 g/l to 600 g/l. They found that there is the increasing interface velocity region located between the first and the second decreasing rate periods. Just as the first falling rate period can be attributed to the propagation of higher concentration layers to the sludge interface as described by Kynch theory, the second increasing rate period can be explained by the propagation of lower concentration regions created by the compression of the deep solids and expulsion of fluid from the compression region. The authors also commented on the presence of channels that move upward as a zone until they meet the descending interface. The channels could not be the cause of the drop in solids concentration because the decreases in concentration occurred prior to the formation of visible channels. The formation of visible channels was observed only in a certain concentration (for calcium carbonate channels occurred across a consistent concentration of about 325 g/ l). It is likely that the lower concentration is a necessary condition for the occurrence of channels.

Bhargava and Rajagopal (1993) using the experimental data, developed models for predicting the first -order rate constants k_t and k_c for both the transition zone and compression zone. The rate constants can be related to the initial suspended solids concentration and sludge volume index. The ratio (k_t /k_c) generally decreased with increase in concentration values for all types of suspended materials tested (chemically flocculated and clay slurry types). The model developed can be used to determine the rate constant k_c for the stated suspended materials for any given concentration X_0 .

According to Cacossa and Vaccari (1994) both theoretically and experimentally identified inadequacies associated with the Coe and Clevenger (1916) gravity thickening model have become apparent: the theory cannot predict the existence of a sludge blanket in an underloaded thickener and it does not account for the final distribution of the concentration in a settled sludge. These behaviors may be accounted for by including the interparticle compressive stress. The authors calibrated the Kos (1978) compressive gravity thickening model with a single batch settling curve. Their model based on force and mass balance was capable of describing both batch and continuous thickening. According to the authors batch setttling experiments for fitting the compressive thickening model should be conducted in columns with the minimum depth of 120 cm, and the minimum diameter of 10 cm.

Regimes for zone settling of the waste activated sludge have been identified experimentally by Chen et al. (1996). Two sludges were tested at several different solids concentrations. For sludges with a concentration above approximately 0.6% the settling curves were ”standard” including the induction period (mostly lasting for only 3-5 min), the constant rate period, the falling rate period and the compression period. When the solids concentration was further reduced (below 0.35-0.5 %) the settling had a different characteristic. After the short induction period (1-2 min) it entered the constant - rate period and then the speed - up period followed than by usually observed falling- rate period and the compression period. The settling velocity at the speed-up period was found higher that in the constant-rate period. The existence of a speed-up phase is explained by creation of large aggregates from an originally macroscopically homogenous sludge. The paper provides also empirical criteria for differentiating settling regimes.

Renko (1996) developed a model for describing sludge blanket interface settling in a batch reactor. In this model the sludge settling curve is considered as one entity not as a conglomerate of several separate parts and thus described in one model. The model interprets sludge settling as a time dependent phenomenon and it describes the settling process from the beginning to the end. Since the velocity is a rate of change it can be computed as a derivative of the sludge blanket level h_t as follows:

$$dh_t/dt = -\alpha\, h_t/X\, h_0 + C \qquad (29)$$

where: h_t = the sludge blanket interface level at time t, m; h_0 = the initial sludge blanket interface, m; X- solids concentration, kg/m^3; C, α parameters describing sludge settling.

The model can be used in modeling the sludge settling curve and the parameters as sludge indexes. It also may be used in writing the graphical solutions of the Kynch theory of sedimentation (Kynch, 1952) and Talmadge's method (Talmadge and Fitch, 1955) to determine the clarifier area in a mathematical form. Since the proposed model describes the whole sludge settling process, it opens possibilities to further modeling of operation of secondary clarifiers.

5. Modeling of clarification and thickening functions

5.1. General background

5.1.1. Batch settling

Most of the recent approaches involve attempts to determine the capacity limiting concentration from the results of a single batch settling tests. Kynch (1952) performed a mathematical analysis of the thickening operations based on the assumptions that the velocity of fall of the particles in dispersion is a function of the local concentration of the particles only. He wrote a continuity equation describing solids entering and leaving an infinitely thin element at the surface of a layer, proved that concentration layers traveled upwards at a constant rate and developed the nature of the subsidence curve for the liquid - solids interface. His batch settling curve can be interpreted to represent the successive intersection of slower-subsiding more-concentrated layers with the interface. The interface concentration at the point -a- is found to be:

$$X_a = H_0\, X_0 / H_z \qquad (30)$$

where: X_a - interface concentration of layer -a-, mg/l; X_0 - initial concentration, mg/l; H_0 - initial height, m; H_a- intercept on vertical axis of tangent to settling curve H(t) at point -a, m;
and also:

$$G = H_0\, X_0 / t_z \qquad (31)$$

where: G- solids flux, kg/m^2 h; t_a- time to point -a- on settling curve. The sedimentation velocity v_i of the interface is determined by the formula:

$$v_i = - dH_i/dt \qquad (32)$$

Using both formulas (29) and (31) it is possible to determine $v_i = f(X_i)$.

This relationship is interesting because it offers a way to derive all zone settling information needed for thickener design from a single settling curve, rather than from a whole series of tests as required by the Coe and Clevenger procedure.

Talmage and Fitch (1955) showed that according to the Kynch analysis, multiple-batch settling tests for determining the limiting solids handling capacity were unnecessary. Since all layers with less capacity than the layer above are ultimately propagated to the surface, their settling rates may be determined from the slope of the interface - time curve. They also developed a geometric construction method for using a single-batch settling curve to establish the area required for an arbitrarily selected rate-limiting layer. This technique, because of its simplicity, has received a considerable attention but later on many authors, including Fitch (1962), have reported that better results are obtained from multiple - batch settling tests. With respect to the effect of suspension compression resistance on the accuracy of the results, the Coe and Clevenger procedure is preferable to that of Talmage and Fitch (Dixon, 1982).

On the basis of Kynch´s and Talmage´s and Fitch´s theoretical discussion a new, very simple graphical method was demonstrated by Jernqvist (1965). The method gave the settling rate as a function of concentration and the flow rate of solid particles per unit area as a function of concentration, directly from batch settling tests. The author developed new theoretical guidelines for the analysis of operating conditions of continuous thickeners including the unsteady state. Experimental batch tests with $CaCO_3$ slurries of different concentrations lend support to this new approach. The settling velocity depend also on the interparticle or compressive force which had not been included into Kynch´s theory. That is why his theories of sedimentation were reinterpreted, modified and extended to be valid for batch sedimentation in which a zone of compacting sediment forms at the bottom of the column. In the compaction zone settling velocity is not dependent on concentration alone, but also on the solids stress gradient.

Michaels and Bolger (1962) developed a model of compressive batch thickening. In their theoretical model, they postulated that the solids floc network in compression would have a compressive yield value σ, which is a function of concentration, and a permeability K that, after an initial period during which floc structure and channeling develop, is also a function of concentration. That is $\sigma = \sigma(X)$, K = K (X), and K in compression is augmented by channeling. They found that the maximum sedimentation velocity diminished linearly with the reciprocal of the initial height. The model has been recognized as analogous to a soil consolidation model and good agreement was found between the theory and experimental results with flocculated kaolin suspensions.

Another solution of the compression hypothesis was given by Kos (1977, 1978). He used the general form of the force balance and calibrated the model, by directly measuring dynamic pressure P, solids stress σ and floc density, ρ_s. The essential difference between the Michaels and Bolger model and the Kos models is that in the former, the settling velocity varies linearly with concentration gradient at any particular concentration, while the Kos model may take nonlinear forms. The compressive yield value in the Kos model is a function of the concentration but the settling velocity is a function of both concentration and dynamic pressure gradient.

$$v = v\,(C, \delta p/\delta x) \tag{33}$$

These methods have not been used for further research perhaps owing to the difficulty of their application.

Further elaboration on the Michaels and Bolger model and Kos´s model was done by Vaccari and Uchrin (1989). They presented a mathematical model of compressive thickening based on Kos´s model that can be calibrated from measurements of concentration, concentration gradient and bulk underflow velocity in continuous thickening experiments. The model was calibrated and tested using steady state continuous thickening experiments with activated sludge.

Another important factor in explanation of deviations from Kynch's sedimentation theories is channeling. In Kynch's theories it is assumed that the particle concentration is uniform across any horizontal layer. Many investigators have observed channels in sedimentation experiments. In formation of channels the hydrodynamical forces are diminished which causes a higher sedimentation rate at a constant concentration.

Fitch (1975) describes different mechanisms that have been proposed in order to explain the formation of channels in sedimentation. They are vertical discontinuity mechanism (Michaels and Bolger, 1962) and horizontal discontinuity mechanism

(Fitch, 1966b). The former one can be explained by rearrangement of interstitial spaces remaining after initial aggregation to form a vertical shell and tube configuration. Horizontal discontinuity mechanism derived from flux continuity and Kynch theory postulates that horizontal pockets of either fluid or suspension of lowered concentration will tend to segregate where the second derivative of the flux curve is positive. At present the evidence seems to favor the vertical discontinuity.

So far there is no theoretically sound and empirically proven procedure that would be fully reliable for the design of thickeners. Many works have been dedicated to this goal (Fitch, 1983,1993; Tiller, 1981) but each of them have deficiencies derive from assumptions not valid in the real world of thickening.

5.1.2. Continuous settling

The first understanding of the thickening process can be found in work conducted by Coe and Clevenger (1916). Working with metallurgical slimes they introduced the concept that each concentration of a suspension has a certain capacity to discharge its solids as given by:

$$G = v/\ (1/X_i - 1/X_u\ ) \qquad (34)$$

where: G = the flux of solids moving towards the underflow at any level, X = solids concentration, v = settling velocity at this concentration, X_u= concentration in thickener underflow.

Coe and Clevenger noted that if a layer in a suspension has a lower solids-handling capacity then the overlaying layer, it will not be able to discharge solids as fast as they are received, and will necessarily increase in thickness. It gave basis for determining the area required for the thickening function of a settling tank that is to provide a sufficient area to assure that solids are applied at rate less than the solids capacity of the limiting layer. The concentration of the limiting layer is called the critical concentration. If solids feed is greater than this, a critical zone of concentration X_c will form in the thickener. As solids cannot pass through it as fast as they are being supplied its height will increase until it fills the thickener and the excess solids overflow.

Coe and Clevenger developed procedures for determining the relationship between settling velocity and concentration from the results of batch settling studies. Several separate settling tests are required in their recommended procedure, which involves experimental determination of initial settling velocity at several concentrations between the influent and desired underflow concentrations.

Similar reasoning, arriving at equivalent equations and making the same interpretation of empirical data was presented by Yoshioka et al. (1957). These researchers recognized that the downward velocity of solids in a continuos thickener was the sum of two components: settling velocity -u- of solids, and a transport velocity - v- of the suspension resulting from underflow withdrawal.
Therefore solids flux:

$$G = X\,(u + v) \qquad (35)$$

Since from mass balance $u = Q_u /A = GA/ X_u A = G/X_u$, rearranging the equation the original Coe and Clevenger equation can be derived. Yoshioka expressed his equation graphically and such constructions were presented frequently in current literature. They make it possible to find the minimum flux by drawing a line from the desired underflow concentration X_u, tangent to the underside of the flux curve. This line will intersect the limiting flux G_c at the Y-axis.

5.1.3. Thickening of activated sludge

Numerous investigators have shown that prevailing thickening theories cannot be strictly applied to activated sludge. The third force, interparticle or compressive force, apart from gravity and viscous fluid drag is particularly significant in biological and chemical sludges.

Dick and Ewing (1967) have described in detail thickening under steady-state conditions. They concluded that interparticle forces probably account for the deviation from prevailing theory and the extend of the deviation can be evaluated by the retardation factor (extend of the deviation from „ideal behavior") and the ultimate settling velocity. The settling of the activated sludge cannot be predicted by the Kynch theory since in addition to being dependent upon concentration the rate of subsidence is dependent upon sludge depth and mixing of underlaying layers. The retardation factor of activated sludges, within the concentration ranges investigated, varied exponentially with sludge concentration; low retardation factors were associated with sludges of good settleability, while bulking sludges had high retardation factors.

Similar results were also reported by Shannon and Tory (1965) and Fitch (1962), who found that the settling velocity of flocculant suspension depends on the conditions under which the aggregates were formed. Aggregates formed by agglomeration of a suspension at a specific initial concentration X_i, may be different than the aggregates formed when a dilute suspension with an initial concentration less than X_i thickness to the concentration X_i.

Javaheri and Dick (1968) provided interesting information regarding the fundamental physical behavior of activated sludge during thickening. They determine the nature of liquid displacement from aggregate particles and the manner in which aggregate particle size varies as consolidation takes place. The water content and size of the activated sludge aggregates have been analyzed by use of the Richardson and Zaki equation (Richardson and Zaki, 1954). Their theory assumes that the initial settling velocity for a swarm of uniformly-sized, nonporous particles can be well correlated by the following expression:

$$V_s = V_0\, \varepsilon^{4.65} \tag{36}$$

where V_s, ε, and V_0 are respectively the settling velocity, the sludge porosity (the fraction of space not occupied by the flocs) and the intercept velocity at ε = 1. The water content of aggregates also has been analyzed according to the equation for flow through porous media developed by Carmen and Kozeny (1937). After analysis of the experimental data the following conclusion could be drawn:

- The aggregates that comprise the sludge are squeezed to eliminate water and are split into smaller aggregates.
- The fluid eliminated from subsiding sludge masses originates from within

aggregate particles and from the interstices between aggregates.

Javaheri and Dick (1968) modified the above equation to yield:

$$V_s = V_0\,(-\Phi_a)^{4.65} \qquad (37)$$

where: Φ_a= aggregate volume fraction. This modification is based on the fact that the suspension porosity ε is that space occupied by free water outside of the flocs. The Aggregate Volume Index (AVI) was introduced as the ratio of volume occupied by sludge aggregates to that occupied by the dry sludge solids alone. Activated sludges with good settling characteristics have low AVI and high porosity values, and thickening to high concentrations occur primarily by elimination of internal water. Sludges with poor settling properties have high AVI values and low porosities. That is, much of the water removed in the course of thickening of poor settling sludges comes from inside the aggregates.

Deviations from the Kynch theory in thickening sludges from waste water treatment plants were investigated by Hultman and Hultgren (1980). They developed a modified consolidation theory to describe the thickening process of activated sludge. In the consolidation model the permeability K, and the compression module M are properties characterizing the sludge. The permeability was a function of the local concentration and varied slightly with time while the compression module was approximately independent of the concentration and time. Based on the studies of the height of the interface as a function of sedimentation time at different initial values of height and concentration the authors developed an empirical model which describes the deviations from the Kynch's theory.

Experiments conducted by Knocke (1986) in both lab and full scales showed a strong correlation between SVI and AVI. Changes in the SRT of the system resulted in significant changes in thickening characteristics. The improvements seemed to be related to an increase in the floc size. Variations in SRT did not seem to have a significant impact on the water content of activated sludge flocs.

5.2. Solids flux theory

Research by various investigators has lead to development and general acceptance of setttling flux theory describing the solids thickening function of secondary clarifier. The theory assumes that the thickening capacity of the secondary clarifier is limited by the values of the mixed liquor MLSS and the return sludge concentration as well as the sludge settling characteristics. At a sufficiently high solids load the capacity is limited by the minimum solids flux. The total solids flux is the sum of the solids flux due to settling and the solids flux downward movement due to sludge removal (sludge underflow).

The total solids flux in a continuous settler at any level between the sludge - supernatant interface and the bottom of the settler can now be calculated as:

$$G_{t.} = G_B + G_U = X\,v + X\,u \qquad (38)$$

where: X = suspended solids concentration, g/m^3 , v = settling velocity of the activated sludge, m/h; u = downward velocity caused by sludge withdrawal, m/h; G_B = batch solids flux, kg/m^2 h, G_U = underflow solids flux, kg/m^2 h; G_T = total solids flux, kg/m^2 h.

The general procedure for determining the maximum available solids loading rate (the limiting solids flux G_l) to a secondary clarifier begins with the development of a plot of the subsidence flux (equal to vX) as a function of X using the v versus X relationship determined for the activated sludge. The curve asymptotes to a line represents the mass flux due to sludge withdrawal alone and it may be observed that there is a minimum in this curve. This represents the maximum value of the solids loading which may be applied to a tank before solids rise and eventually flow into the effluent. If the applied flux is greater than the limiting flux, then the sludge blanket will increase, and if the applied flux is less than the limiting flux then the sludge blanket will decrease. According to solids flux theory, thickening failure results when G_T exceeds G_L (the limiting solids flux).

This is evident because although at the feed concentration the rate of mass flow per unit area, and hence the applied solids loading, may be quite high, as the solids fall through the tank and become more concentrated. The available rate of mass flow per unit area will therefore decrease and at some point in the secondary clarifier there will be a critical concentration at which the mass flux is a minimum. As the sludge concentration usually has to pass through the critical concentration to achieve the underflow concentration, it is this critical concentration which defines the maximum solids loading. The design of a secondary clarifier is based essentially on estimation of the required area for that value of the limiting flux, G_l .

Waters and Galvin (1991) presented a new experimental procedure for determining the solids flux curve. The procedure is based on measuring the concentration at various heights of a bed of settled solids formed during a semi-continuous sedimentation test. The advantage of this method was that the solids flux curve could be determined at low and high feed fluxes. Unlike conventional thickener area calculations the test procedure demonstrates the dependence of the flux curve on the system feed flux. At very high fluxes the resultant flux curve approached that obtained by the Kynch method. This was in an agreement with the observations that the effective feed fluxes of the batch tests used in the Kynch method were always relatively high. The Coe and Clevenger method was found to be invalid for all feed fluxes.

The importance of the flux theory in settling tank design and operation was first established by Coe and Clevenger (1916) and Kynch (1952). These early works were performed on unflocculated suspensions. Although activated sludge is a flocculated suspension it has been shown that the mass - flux concept also can be applied to activated sludge (Dick, 1970;1972). Now, developed and well demonstrated solid-flux theory quite well predicts the thickening performance of full- scale suspended growth process clarifiers that are not limited by their capability to thicken the sludge. While mass flux theory provides a rational basis for the selection of surface area and underflow rates in its present form it has some limitations prediction of the concentration of solids in the underflow. (Rachwal, 1982). One constrain in the application of this theory has been difficulty in developing the required initial settling velocity (v) versus initial suspended solids concentration X relationship for operating the system.

5.3. Settling velocity models

Application of the mass flux concept to the settling tank behavior has been particularly assisted by the graphical method of Yoshioka et al. (1957). This method was developed from multiple batch settling tests for the purpose of obtaining the settling velocity as a function of concentration, upon which the criteria for settling tank area for the thickening function (solids transfer to the bottom of the tank) is based.

Despite developments in the graphical flux theory the procedure remains rather tedious, as for each concentration the settling velocity versus sludge concentration curve needs to be constructed to estimate the settling velocity. In order to overcome the difficulties associated with the graphical representation of the flux theory considerable research has been undertaken to find an empirical relationship between the settling velocity and sludge concentration. This relationship allows direct analysis of settling tank behavior.

Many mathematical relationships, both theoretical and empirical, between the settling velocity -v- and the concentration X_o have been proposed. Some of them are presented in Table 9 (Hultman et al. 1991). Acceptance of one or the other of these formulas in the flux theory leads to significant differences in the interpretations of the behavior of settling tanks. The constant parameters in the formulas can be obtained experimentally e.g. in the Vesilind formula the parameters -V_0 - and -n- can be easily determined by doing sedimentation tests with the sludge at different concentrations. Settling velocity is determined from the linear part of the sedimentation curve. Log v is then drawn against sludge concentration and V_0 and -n- are determined from the plot using linear least-squares regression of log v against concentration X over a range of sludge concentration from 1 to 12 g/l.

The two most widely accepted formulas are those of Vesilind (1968) and Dick and Young (1972). It should be recognized that none of these models accounts for compression effects and channeling effects at the bottom of the clarifier as described by Vesilind (1979). Smollen and Ekama (1984) has conducted a comparison study of these two mathematical expressions and described the differences between the predicted behavioral patterns. They found that theoretically the Vesilind´s formula yielded a more constant and intuitively satisfying flux theory for secondary clarifiers. Statistical evaluation of 159 sets of experimental data (from the writers and literature) indicated that the Vesilind´s formula gave a better correlation than the Dick´s formula.

Table 9 Relationships between sludge settling velocity and concentration

Formula No.	Formula	Investigations of the formula
1	$v = V_0 - nX_o$	Kalinske (1948), Bond (1961)
2	$v = V_0(1 - nX_o^{2/3})$	Bond (1961), Brown and LaMotta (1971)
3	$v = V_0(1 - nX_o)^{4.65}$	Richardson and Zaki (1954), Michaels and Bolger (1962),
4	$v = V_0 e^{-n X}$	Vesilind (1968)
5	$v = V_0 X_o^{-n}$	Yoshioka et al. (1957), Dick and Young (1972)
6	$v = V_0 e^{-n X}/X_o$	Cho et al. (1993)
7	$v = V_0 (1 - nX_o)^4/X_o$	Cho et al. (1993)

where: v - settling velocity, m/h; X_0 - MLSS concentration, g/m^3; V_0 (m/h), n (m^3/kg MLSS or l/g) - constants describing sludge characteristics.

Also Hultman and Li (1995) have evaluated the settling velocity formulas and found that Vesilind`s and Dick and Young`s formulas can be recommended above others because they are widely used which means that the obtained coefficients in the formulas can be compared with other experimental studies. This is important because the used values of coefficients are crucial for the results. On the other hand the other formulas may for some cases give much simpler relationships and it may sometimes be advantageous to relate the settling model with the characteristic parameters to explain the settling behavior.

5.4. Influence of sludge indices on solids flux

Design based on the thickening criterion accomplished through the application of solids flux theory has not been widely adopted in practice because it is based on the experimental development of a settling flux curve which characterizes sludge settleability. Since this is both time consuming and labor intensive such curves are not routinely developed and used by design engineers. Moreover, the extended periods of settling (2-3h) required at the higher concentrations (over 6 g/l) may lead to denitrification (if sludge nitrifies) that causes severe retardation of ZSV and some time flotation.

Recognizing the need for a more easily applied approach for designing and analyzing the operational state of secondary clarifier several recent investigations have been reported in which the sludge indices like SVI or several of its modifications have been empirically related to the two settling parameters -V_0 - and -n- of the Vesilind equation. The results suggest that there should be a single empirical relationship between the initial sludge setttling velocity (v_i), the initial sludge concentration (X_i) and the sludge settling parameters (SVI, SSVI, DSVI) that can be used to accurately predict the settling velocity of a wide variety of activated sludges. Such relationships would be helpful in developing broadly applicable design and operation charts for activated sludge secondary clarifiers. It could be used for designing a secondary clarifier according to the thickening criterion and evaluating various economic trade-offs to determine a cost-effective design. It would also give ability to analyze and assess the operation of an activated sludge system and formulate operational control strategies to properly manage sludge inventory.

5.4.1. Foster Model

The model developed by Foster (1982) used other than SVI parameter for describing the sludge settleability. The Stirred Sludge Volume Index (SSVI) was suggested for developing a relationship between Vo, n and sludge settling characteristics. The work conducted by Foster showed that parameter n varied smoothly with the SSVI and the settling velocity V (m/h) could be calculated from the SSVI (ml/g) and concentration X (mg/l) using the equation :

$$V = 5\, e^{-[0.2498\, \exp(0.0046 SSVI)]X} \qquad (39)$$

5.4.2. Pitman, Ekama and Marais Model

Pitman (1984) studied the settling properties of activated sludge from four biological nutrient removal plants. Generally, well- settling sludges had high V_0 values around

13 m/h and low n values around 0.25 m^3/kg, whereas poorly settling sludges had low V_0 values around 5 m/h and high n values around 0.5 m^3/kg. He found that V_0 increased and n decreased as settling properties improved but the correlation coefficients of V_o and n versus SSVI and SVI were rather low. But when V_0 and n were expressed as ratio V_0/n good correlation were obtained. The correlation between V_0/n and SSVI was better than the correlation between V_0/n and SVI. The relationship he obtained was:

$$V_o/n = 67.9\ e^{(-0.016SSVI)} \qquad (40)$$

To verify the data by Pitman, Ekama and Marais (1986) compared them with the data reported in the literature by Rachwal (1982) and White (1975). The data show a remarkable close correlation to the data by Pitman. Knowing V_0/n from the SSVI did not define V_0 and n individually. This problem could be overcome because White's, Pitman's and Rachwal's data indicate that as V/n increases (i.e. settleability improves) then -n- decreases. All three sets of data correlate well if plotted -n- vs. log (V_0 /n). From Pitman's data:

$$n = 0.88 - 0.393 \log(V_0/n) \qquad (41)$$
$$(r^2 = 0.976)$$

Once -n- is known, V_0 can be found from the equation:

$$V_0 = (V_0/n)\ n \qquad (42)$$

Pitman, Ekama and Marais consolidated the WRC and solids flux theory into a single approach so with the use of only SSVI data the design engineer and plant operator has at his disposal both the WRC and flux procedures and can utilize aspects of both in his work.

5.4.3. Daigger and Roper Model

Daigger and Roper (1985) developed an empirical correlation between SVI of an activated sludge and its settling characteristics as represented by the relationship between v and X. The authors presented a practical model for estimating activated sludge settling characteristics on 46 separate activated sludge samples with SVI ranging from 36 to 402 ml/g. A total of 236 individual settling velocities were measured and the results were fitted to the model:

$$V = V_o e^{-n X} \quad (43)$$

The model parameter V_0 was found to be independent of SVI with a mean value of 7.80 m/h. The other parameter n was highly dependent on SVI and increased linearly with SVI. The following relationship was determined:

$$V = 7.8\, e^{-[0.148 + 0.00210(SVI)]X} \quad (44)$$

where: 36 ml/g< SVI < 402 ml/g

Based on the solids flux theory the authors developed an operational diagram relating the solids loading rate to the clarifier with the return activated sludge (RAS) solids concentration.

The model was developed based on experimental multiple batch settling tests conducted at two full -scale and six pilot scale activated sludge wastewater treatment plants receiving only two different influents treated with pickle liquor for phosphate removal. Consequently, it is clear that although the general concept of the chart developed and presented by Daigger and Roper (1985) is excellent and is extremely useful for pickle liquor amended phosphorus removal plants, application of the specific chart may lead to nonconservative secondary clarifier designs and should be carefully used in operating activated sludge systems without chemical phosphorus removal.

5.4.4. Wahlberg and Keinath Model

The study of Wahlberg and Keinath (1988) focused on the development of a functional relationship between the SVI measured according to four different techniques of determining SVI and the two adjustable parameters, -V_0 - and -n- of the

Vesilind formula for describing interface settling velocities as a function of concentration X. The authors based on their results obtained from 21 full- scale activated sludge plants that varied considerably with respect to size, geographic location, mode of operation, method of aeration and type and amount of industrial input. None of the sludges tested were chemically amended.
The authors developed the empirical model for predicting settling flux from SSVI data:

$$v = (153 - 0.615\ SVI)\ \exp - 80.426 - 3.84*10^{-3}\ SVI + 5.43*10^{-5}\ (SVI)^2 X$$
(45)

where: 35 ml/g < stirred SVI < 220 ml/g

Statistical identification of the behavior of the relationship between the settling parameters and SSVI indicated that there is a linear relationship between V_0 and SSVI and a parabolic relationship between n and SSVI. Traditional SVI was found to be a very imprecise measurement. The results obtained during the study have shown that there is an excellent agreement between the model and the actual data obtained from the operating plants.

Further exploring this concept Keinath (1990) developed a design and operation chart for use by design engineers and wastewater plant operators based on the batch settling data collected at 21 full scale wastewater treatment plants. The diagrams related the solids loading rate to a functional group containing MLSS concentration and the recycle ratio as a main parameter. Hermanowicz (1998) using Daigger and Roper (1985) and Wahlberg and Keinath (1988) relationships presented another operating diagrams that connected the process operational parameters: MLSS, return activated sludge solids, clarifier hydraulic loading and recycle ratio in a graphical form.

5.4.5. Cho Model

Another approach for determination of an appropriate settling velocity model to be used for calculation of the settler system with the limit solids flux theory was presented by Cho et al (1993). The model starts with the Carmen - Kozeny equation which is developed into new models by adding the slurry viscosity term. They are:

$$v = V_0\, e^{-nX} / X, \tag{46}$$

$$v = V_0\, (1-nX)^4 / X, \tag{47}$$

$$v = V_0\, (1-n_1 X)^4\, e^{-n_2 X} / X \tag{48}$$

where: X= solids concentrations, V_0, n, n_1 and n_2 - constants

The power model, the exponential model and the new model v = k exp (-nX)/C fitted well with the experimental data and the last fit was the best. The new model is suggested to be used for calculations of activated sludge secondary settlers instead of the power model which is not usable in the dilute concentration range and instead of the exponential model which is complicated to be used with the limit flux theory

5.4.6. Daigger Model

Further elaboration on the this type of relationship was presented by Daigger (1995). He summarized an extensive sludge settling database consisting of data collected by five different sets of researches: Daigger and Roper (1985); Pitman, (1984); Wahlberg and Keinath, (1988);Tuntoolavest and Grady, (1982) and the data, that has not been previously published provided by George Ekama of the University of Cape Town. All together the data represented 51 separate activated sludge biological wastewater treatment processes, including 214 separate runs and over 1 500 individual data points. The author attempted to present the most representative secondary clarifier operating diagram through pooling the currently available data. The results indicated that the wide variety of data could be accurately described by a single relationship as:

$$\ln v = 1.871 - (0.1646 + 0.00158\ SVI)\ X \tag{49}$$

The predictive capability of the relationship was confirmed using an independent data set. This relationship was further used to develop secondary clarifier operating diagrams. The predictive capability of three sludge settling indices was considered in this research (SVI, DSVI and $SSVI_{3.5}$). It was found that predictive capability of

$SSVI_{3.5}$ is much better then two other indexes considered. While it is hoped that the use of the conventional SVI test can be eventually eliminated from practice it seems that the diagrams based on it can be still used with reasonable accuracy to predict the thickening characteristics of activated sludge because of the extensive database to develop it. Additional data should be collected to compare other sludge setttling indexes (DSVI and $SSVI_{3.5}$) and to develop refined empirical characteristic correlations for these indexes.

It seems that activated sludge plant type is a factor that might cover a number of subfactors such as temperature, filamentous organisms types, biological nutrient removal, completely aerobic, mixing intensity and turbulence (Rasmussen and Larsen, 1996). Therefore, use of any relationships presented above should be restricted as far as possible to the type of activated sludge plants on which the data set was measured.

5.5. Clarification and Thickening Models

5.5.1. Settling and thickening zone

A predictive model describing the secondary clarifier efficiency based on fundamental mechanisms has not been developed for a very long time because of the complexity in modeling process. Today there are plenty of existing models for the secondary clarifiers performance and their complexity range from a very simple empirical models to some very complicated ones. In some cases they have been combined to represent the entire activated sludge system and predict the distribution of sludge solids between the reactor and the settler.

A considerable amount of research have been undertaken to study flow patterns and hydraulics of clarifiers. Price and Clements (1974), Ostendorf (1986), McCorquodale et al. (1988), Bretscher et al. (1992) have studied such hydraulic phenomena as: turbulent dispersion and mixing, bottom density current, buoyant density current, short circuiting, recirculation and density waterfalls in the inlet mixing zone (Zhou and McCorquodale, 1992). They all are strictly clarification models and those models have not been discussed in this review. The attention has been focused on models that try to capture the interaction between both setttling and thickening function of the clarifier. Some models often try to extend the flux theory to the discrete setting conditions in the upper effluent region modifying it for low concentrations.

The approaches to extend the flux model to low concentrations can be divided into two categories Ekama et al. (1997):

- Two groups of particles are defined. Most of the particles are treated as sludge and are described according to the flux theory. A small group of single particles is additionally defined with settling velocity and is set as a very low value or even zero. (Dupont and Henze, 1992).
- The basic approach of the flux theory is kept, that is to estimate the settling velocity as a function of the concentration, but modified so that the function starts at zero settling velocity for very low concentrations rather then at some maximum value (Takacs et al. 1991; Dupont and Dahl, 1995).

Both approaches significantly improve the accuracy of the dynamic model prediction of the effluent suspended solids concentration in 1- D and 2 - D models.

George and Keinath (1978) formulated a mathematical model that predicted the dynamic performance of the clarifiers and thickeners under normal operating conditions. Their experiments supported the concept that the limiting flux governs the ultimate transport of solids. Satisfactory agreement was found between the observed and simulated dynamic responses of the underflow solids concentration and the sludge blanket height to various imposed solids flux and underflow withdrawal rates.

Empirical model that predict the clarification performance of the secondary clarifier was developed on a pilot plant scale with activated sludge (Chapman, 1983). It began as a part of a research program to quantify the effects and importance of different design and operating variables on clarifier efficiency. The variables selected for investigation included: MLSS concentration, clarifier feed flow, and underflow rates, the air flow rate to the aeration tanks preceding the clarifier, the depth of the inlet feedwell, the clarifier sidewater depth and the speed of the sludge collection arm. The data were analyzed using regression techniques. The best regression equation contained five terms; expressing flows as hydraulic loading per unit surface area of clarifier the appropriate equation was:

$$X_e = -180.6 + 4.0\ MLSS + 135.6\ Q_a/A + SWD\ (90.2 - 62.5\ Q_a/A) \qquad (50)$$

Where: X_e - effluent suspended solids, mg/m^3, MLSS - concentration of suspended solids in the mixed liquor, g/m^3; Q_a/A - settler feed flow rate per unit of surface area = $Q_i/A + Q_r/A$; Q_i/A - plant inflow per unit of surface area, m/h; Q_r/A - recycle rate per unit of surface area, m/h; SWD - sidewater depth, m.
The equation was applicable only over the range of parameters used in the experiment.

Another approach is found in the work of Dupont and Henze (1987). The authors described the development of a model for the secondary clarifier based on the general flux theory for zone setting, which can be used in combination with a dynamic computer model of the Activated Sludge Model no.1. Previous models included a very simple model for the clarifier which was based on the assumption that the secondary clarifier was always able to concentrate the sludge and it meets the desired sludge age for the plant; sludge age was not calculated but used as a operating parameter. In Dupont and Henze model zone settling model has been used with the Vesilind formula for the expression of the settling velocity (Vesilind, 1979). Moreover, the developed model includes a function that predicts effluent concentration of non settled particles in the effluent. This model is a purely empirical model, which links the effluent quality with the hydraulic load, suspended solids load and the nitrate load. The model might be limited in its application to real world plants because it is based on a pure cylindrical clarifier. Further research on the modeling of the secondary clarifier combined with the Activated Sludge Model no. 1 was presented by the authors in their later paper Dupont and Henze (1992).
Erickson et al. (1991) tried to design an activated sludge system so as to minimize the total of its volume. The secondary clarifier model includes a description of clarification and sludge thickening so that both the overflow and underflow streams can be characterized. The settling velocity of activated sludge has been modeled using the power law model and the linear model depending on sludge concentrations.

$$X<X_l \qquad v_s = k_1 - k_2X \tag{51}$$

$$X>X_l \qquad v_s = k_3\,X^{k4} \tag{52}$$

where: X = sludge concentration, mg/l; X_l = concentration at which settling velocity mode changes, mg/l; k_1, k_2, k_3, k_4 = constants. The value of constant X_l for sludge with poor settling character was 2200 mg/l; for average settling character X_l = 3540 mg/l.

Further development in this area was presented by Akca et al (1993) and their optimization model of the activated sludge treatment plant. The thickening function of the secondary clarifier design is based on the limiting flux theory. Sludge volume index (SVI) which is related to the solids flux constants has been used as a design parameter in this study. A relationship was developed between SVI (ml/g) and sludge age Θ (d) using data from literature. The mathematical expression of this relation was determined as:

$$SVI = 246.9\, e^{(-0.0742\, \Theta)} \qquad (53)$$

In the optimization model the settling velocity was assumed according to the Vesilind´s formula (1968). Having evaluated the published data (Pitman et al., 1983; Pitman, 1985; Daigger and Roper, 1985; Knocke, 1986) the correlation between sludge volume index and the two constants (V_o, n) of the settling velocity equation were also found. They are as follows:

$$V_o = 28.1\, (SVI)^{-0\ 2667} \qquad (54)$$

$$n = 0.177 + 0.0014\, (SVI) \qquad (55)$$

Combining both sets of equations V_o and n can be expressed in terms of sludge age Θ:

$$Vo = 6.466\, \exp(0.0198\, \Theta) \qquad (56)$$

$$n = 0.177 + 0.346\, \exp\,(-0.0742\Theta) \qquad (57)$$

The model suggests that sludge settleability is improved with increased sludge age. It must be noted that this suggestion is limited only to the conventional activated sludge process. The model suggest also that the design dissolved oxygen concentration in the aeration tank may vary with the sludge age; at lower sludge ages DO must be higher while at higher sludge age DO concentration must be lower.

Härtel and Pöpel (1991) has presented a dynamic model simulating the processes of settling and thickening in the secondary clarifier and integrated it into a dynamic biochemical reaction model. The settling velocity is expressed as a function of the solids concentration X and the SVI:

$$v_s = (17{,}4e^{-0{,}0113\ SVI} + 3.931)\ \exp[-(0.9834\ e^{-0.00581\ SVI} + 1{,}043)\ X \qquad (58)$$

The velocity of hindered settling as measured by observing the initial lowering of the sludge interface is adjusted by the mathematical correcting Ω - function to be applicable also for the domain of transition and sludge compression. The settling flux is multiplied by the Ω - function defined as:

$$\Omega = B_1 - B_2\ /1 - B_3\ z^{\ -B\ 4} \qquad (59)$$

where: z- tank depth, m; B - parameters that must be measured or are given by the authors. The validity of the model approach is evaluated by simulation runs showing the velocities of hindered settling, the concentration profiles of MLSS over the tank depth and the effluent concentration of MLSS. Basic variables for the validation are the influent MLSS concentrations and the sludge volume index quantifying the settling characteristics of the sludge. The model requires large computational efforts and the knowledge of many parameters. Including the clarifier model into the integrated models predicts with better results the MLSS concentrations in the activated sludge tank and effluent concentration especially for periods of higher storm water flows to the treatment plant.

Takacs et al. (1991) presented a dynamic 1-D model of the clarification-thickening process. Based on the solids flux concept and on a mass balance around each layer of a one-dimensional settler this model can simulate the solids profile throughout the settling column, including the underflow and effluent suspended solids, concentrations under steady-state and dynamic conditions. The model uses a special settling velocity equation designed to simulate the settling velocity of dilute and more concentrated suspensions since Vesilind's formula applies only to hindered settling conditions. As the solids concentration in the upper layers of the clarifier decreases below the hindered settling concentration, settling velocities predicted by Vesilind's equation will exceed the actual settling velocity of the floc particles as predicted by Li and Ganczarczyk (1987). The model describes settling velocity as the sum of two exponential terms:

$$v_{sj} = v_0 \exp(-r_h X^*_J) - v_0 \exp(-r_p X_j^*) \qquad (60)$$
$$0 \le v_{sj} \le v_0$$

where: v_{sj} - settling velocity of the solids particles in layer j, m/d; v_0 - maximum settling velocity m/d; r_h - settling parameter characteristic of the hindered settling zone, m^3/g; r_p - settling parameter characteristic of low solids concentration, m^3/g; X_j^* - X_j - X_{min}; X_j - suspended solids concentration in layer j, g/m^3; X_{min} - minimum attainable suspended solids concentration, g/m^3; $X_{min} = f_{ns} X_{in}$; X_{in} = mixed liquor suspended solids entering the settler, g/m^3; f_{ns} - non-settleable fraction of X_{in}. The first term reflects the settling velocity of large particles while the second term is a velocity correction factor to account for the smaller slowly settling particles. The model was applied to full scale assuming the steady-state conditions of the experiment, and to the dynamic pilot scale data with very good results.

Another semi-empirical model for determination of activated sludge concentration at the bottom of the secondary clarifier was presented by Roche et al (1995). The model gives a new equation for activated sludge thickening:

$$X_b = a\,(t + \Delta t)^b \qquad (61)$$

$$\Delta t = 0.086\, X_0^{\,2.234} \qquad a = 2.065\, X_0^{\,0.328} \qquad b = 0.545\, SVI^{0.152}$$

where: X_b -biomass concentration at the bottom section of the clarifier, g/l; t - time, min; Δt - compression time advancement parameter, min; X_0 - initial concentration, g/l; a, b, - thickening model parameters. The model is determined from experimental results (23 different experiments) with formulas to calculate a, Δt and b. The formulas have been fitted using settling data obtained on biomass sampled on different activated sludge treatment plants. Model predictions were compared with measured data of a full scale dynamic secondary clarifier with a good agreement.

Dupont and Dahl (1995) presented a dynamic one-dimensional flux model for the secondary settling tank which is suitable for use with the latest innovations in models for activated sludge tanks, and which takes into account observed effects of density current and short circuiting. The components of the influent to the settling tank are divided into three fractions. Soluble components, non-settleable particulate components (primary particles), and settleable particulate components (macroflocs). Soluble components and primary particles are considered to follow the hydraulic flow in the settling tank. The transport of macroflocs in the settling tank is modeled according to the traditional flux theory on layer model of the settling tank extended with a model for density current and short-circuiting. Settling velocities of the macroflocs for both free and hindered settling were measured and a new model for the settling velocity was proposed.

The formulation selected for the description of the settling velocity is the log normal function. It covers both the free settling zone and the hindered settling zone and therefore it has an increasing settling velocity for increasing concentrations at low SS concentrations and a decreasing settling velocity for increasing concentration at high SS concentration. The formula is given by:

$$V_s = V_0 \exp\left(-0.5\left(\frac{\ln\left(\frac{Xss + Xpp}{n1}\right)}{n2}\right)^2\right) \qquad (62)$$

where - V_s - settling velocity for microflocs, m/h; V_0 - maximum settling velocity for macroflocs, m/h; Xss - macrofloc concentration in the settling tank, g/m^3, Xpp - concentration of primary particles in the influent to the settling tank, g /m^3, n1, n2 - sludge characterization constants.

The model was validated with the data from the Lynetten WWTP in Copenhagen, Denmark where the SS concentration profile and the return sludge were predicted well with the model. The authors claim that the new settling tank model is believed to improve models for wastewater treatment plant such as EFOR (1993) due to more precise modeling of the retention time and SS concentration profiles in the settling tank.

Another approach was presented by Ji et al. (1996) in their model which simulated the interaction between the biological reactor and secondary clarifier. The model coupled the model of biological reactor with a two- dimensional hydrodynamic model of the clarifier. Two -dimensional unsteady clarifier model can predict the settling tank performance including effects of density currents and sludge recirculation. The stream function is defined to describe a two- dimensional flow field. The suspended solids transport model is based on the law of conservation for solids in the computed flow field, turbulent diffusion and settling of the flocs. The behaviour of suspended solids in turbulent flow due to convection and turbulent diffusion is solved in the computation of the flow field. The settling velocity, calculated on the basis of the functional relationship between the SVI and the coefficients in Vesilind equation by Wahlberg and Keinath (1988) along with the double exponential formula proposed by Takacs et al. (1991) have been used to estimate the gravitational settling of flocs. The model was verified against the field data from the wastewater treatment plant and showed a good agreement.

The latest model presented by Diehl and Jeppsson (1998) also tries to incorporate but in a separate way both the dynamics of the biological reactor and the dynamics of the settler. Their settler model, that may be combined with the IAWQ AS model, is a first order model that uses the wave behaviour and the conservation of mass. It is based only on the conservation of mass and Kynch's batch settling flux assumptions. The numerical algorithm is derived from non-linear partial differential equations without adding any extra assumptions. The models include also the prediction of the concentrations of all biological components of the particulate and soluble material within and at the outlets of the settling tank.

For simulation of aeration-sedimentation operation a Danish simulation package EFOR (EFOR 1993) can be used. It has models for the reactions in the aeration basin (mainly based on the IAWPRC model) and for the secondary clarifier. The sedimentation model is divided into two different processes. The thickening model is based on the solids flux theory.

Using the models listed above to the design and operation of final clarifiers requires knowledge of experimentally derived constants which describe sludge settling properties. Routinely measured sludge volume indices can not be used in these models. Therefore, formulas and nomographs have been developed to connect the information of sludge settling properties measured as sludge volume indices to the models.

5.5.2. Compression zone height

In continuous thickener design there are mainly two aspects considered: the thickener area per unit of solids flow and the compression zone depth. While the former aspect has been discussed widely in literature the compression zone depth has not received so much interest. There have been some equations proposed in literature (Fitch, 1966; Dixon, 1981) that requires data on the compressibility and permeability of the sediment as a function of solids concentration. However, to obtain the latter data from batch tests it normally requires an extensive experimental work.

Other methods are based on the results of only one batch test for calculation of the compression zone depth in continuous thickeners:

- a method proposed by Coulson et al. (1978) assumes that the time required to concentrate the sediment after it has reached the critical concentration (boundary concentration between settling zone and compression zone) can be determined approximately by allowing a sample of the suspension to settle and measuring the time taken for the interface between the suspension and clear liquid to fall to such a level that the concentrations are reched that are required in the underflow from the thickener.
- Another similar method proposes the calculation of the compression zone height from mean residence time of solids, calculated between the critical point of a batch test and a latter point (Foust et al., 1960).

Font (1990) has presented another approach to calculate the compression zone height in continuous thickeners. With this method it is necessary to know the variations of the sludge - supernatant interface and the sediment height vs. time in a batch test. The method considers the following aspects: relationship between the settling rate of solids and their concentration in the settling zone, and the compression of solids due to the squeeze transmitted by the upper layers - unbuoyed weight of particles minus force of friction due to the Darcian flow. When the variation of the sediment height vs. time becomes linear it is possible to calculate the maximum solids concentration which can be reached by sedimentation. The change of the solids matrix permeability and its influence on the method proposed are also analyzed.

Another approach in optimization of the sedimentation tank operation is prediction of sludge blanket failure. Such a model has been developed by Severin and Poduska (1985). It is based on the solids flux theory. The only data required for the model application is the settling velocity of the MLSS. Once the measured velocity is compared to the predicted minimum velocity required to avoid blanket failure, it is possible to rapidly define the minimum required underflow for any loading condition:

$$V_0 = \frac{1}{A(n-1)}\frac{(Q+U)^n}{U^{n-1}}\left(\frac{n-1}{n}\right)^n \qquad (63)$$

where: V_0 = sludge settling velocity for concentration C_0, m/h; A= clarifier cross-sectional area, m^2; Q = overflow rate m/h;, U = underflow rate, m/h; n = sludge settling parameter,

A rearrangement of the equation above produces a dimensionless expression of clarification parameters which identifies the fundamental dimensionless parameters as: n, $V_0A/(Q+U)$ and $V_0\ A/U$.

$$V_0\ A/(Q+U)= (n-1)^{n-1/n}/n\ (V_0\ A/U)^{n-1/n} \qquad (64)$$

The model predicts the optimum return sludge flow for cost-effective operation. Further mathematical developments led to the ability to predict the rate of rise or fall of sludge blankets of chemical sludge in a laboratory scale clarifier (Severin and Poduska, 1986) and in a full- scale activated sludge plant treating industrial wastewater (Severin, 1991).

Conclusions

1. Successful operation of secondary clarifiers in the wastewater treatment system requires consideration of three basic functions: clarification, sludge thickening and sludge storage. All those function can be satisfactory accomplished when determination of sludge settling behaviour, reliable measurements methods for establishing sludge settling qualities, and application of process design procedures based on flocculation and sludge settleability will be employed in a secondary clarifier concept.

2. The settleability of activated sludge is affected by the composition and characteristics of its microorganisms. Attempts were made to relate sludge settleability to fundamental observations of microorganism type and surface chemistry characteristics. In spite of an extensive research work in this area and better understanding of the relationship between fundamental characteristics and settleability it is still necessary to determine how these fundamentals are affected by design, operating conditions and wastewater characteristics.

3. Though identification of activated sludge microorganisms is one of the most difficult tasks there is no need to identify all microorganisms. For carbon removal microorganisms or nitrifiers the level of knowledge is quite satisfactory but poly-P bacteria and filamentous organisms still require more research. Nocardioform bacteria which look very similar in mixed liquor or foam samples under the microscope, differ widely in their taxonomy, physiology and biochemistry in pure culture. These variations are reported also for other filamentous organisms. Therefore it now appears to be an urgent need for a large research program directed toward taxonomic work, using molecular techniques to resolve how different these filament morphological types are since considerable differences between isolates from different locations do exist.

4. Better understanding of bulking phenomena and the importance of the environmental factors on the bacterial competition in stressed by many authors. Defining the conditions for biological selection of floc forming microorganisms seems

to be the most rational approach for controlling an excessive growth of filamentous organisms. Although the experimental investigations have produced a deeper understanding of the low F/M filament behaviour in N and N + P removal systems a lot more research is required before low F/M filament proliferation can be effectively controlled in full scale N and N + P removal plants.

5. The fate of the particulate fraction (including dissolved complex organic compounds) supplied in the influent plays an important role in the competition between floc forming and filamentous species in nutrient removal plants. The role of particulate substrates in bulking should be clearly defined, especially the rate of hydrolysis under different electron acceptors and the access of filamentous microorganisms to the hydrolysis products.

6. Properties of wastewater flocs such as strength, density, settleability, filterability and water binding have been studied empirically, but systematic studies on the relationship between such properties and the basic constituents of the flocs, the ECP, have not been completed. Further research should focus on the relationship between ECP composition and variation in its characteristics and floc properties; mainly on the role of ECP in floc formation under various environmental conditions.

7. Good measurement parameters for sludge settleability are still required though DSVI and SSVI need to be universally adopted to replace the popular but poor SVI. Relationships between sludge settleability parameters SVI, DSVI and SSVI and flux constants V_0 and -n- have been found to be inconsistent between some data sets and seem to reflect different types of activated sludge plant operating conditions. More research should be directed toward acquiring more information about the effect of activated sludge plant operating condition on the sludge settleability parameters.

8. Current approaches for extending the flux theory to include the low concentration range are inconsistent with the understanding of the settling behaviour of discrete particles. More close reflection of real settling that occurs in discrete non-flocculent and discrete flocculent zones of the secondary clarifier should be developed and incorporated into existing clarification models.

9. There are plenty of existing models for the secondary clarifies performance

and their complexity range from a very simple empirical models to some very complicated ones. In some cases they have been combined to represent the entire activated sludge system and predict the distribution of sludge solids between the reactor and the settler. However, the integration remains incomplete since the models require that the settling characteristic of the sludge is specified. These characteristic are typically determined through a series of batch settling tests or simpler procedures like the SVI to which other measures of settleability have been correlated. Such methods can only be applied for a limited period of time and cannot be applied to the other systems. A completely integrated model of the activated sludge system will only be possible when the settling characteristics of the MLSS can be determined within the model.

10. Application of the solids flux procedure to full scale rectangular and circular secondary clarifiers indicate that the flux procedure overpredicts the permissible solids loading by about 25%. The estabilishment of the applicability of the flux procedure to circular and rectangular clarifiers should be a high priority research. Recently developed hydrodynamic models that allow a measure of tank geometry and internal layout optimization may facilitate calculations of the solids loading rate capacity of different clarifiers designs and layouts and compare these with the flux procedure predictions.

11. Consolidation behaviour of the sludge in the bottom of the secondary clarifier should receive more research attention so as the process could be realistically modeled. Sludge compaction seems to limit the attainable underflow concentration and there is a need for more comprehensive equations describing the sludge layer. Modeling of the clarifiers should also include the effect of turbulence and internal density waves on the sludge blanket.

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