

# Dynamic modelling of nutrient deficient wastewater treatment processes



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## Summary

Being one of the major consumers of natural resources (wood, water) and energy (fossil fuels, electricity), pulp and paper production leads to significant impacts on the environment. In terms of freshwater withdrawal, the pulp and paper industry ranks third in the world, after the primary metals and the chemical industries (Thompson *et al.*, 2001). If not efficiently treated, the effluent mill water will contain large amounts of organic matter and/or nutrients promoting microbial growth, eutrophication and oxygen consumption in the recipient.

Nitrogen and phosphorus supplements are generally required for efficient biological treatment of pulp and paper wastewater. One of the main challenges in controlling the process is to add sufficient amounts of nutrients to obtain high COD (chemical oxygen demand) reduction without releasing the additives into the recipient. The characteristics of biological systems, such as long time-constants and variable nutrient uptake, imply that over- or under-dosage is not immediately detected in residual effluent nutrient concentrations. This makes the implementation of feedback control difficult and a dynamic mathematical model becomes an important tool for the development of robust control strategies.

Prior to the development of a satisfactory nutrient dosage control strategy, knowledge about the biological response to conditions with low and transient nutrient concentrations must be obtained. Based on this information a mathematical model of the system is developed, which is important for control system design. Several mathematical models are available describing COD removal processes, for example the Activated Sludge Model No. 1 (ASM1) (Henze *et al.*, 2000). They have, however, primarily been adapted to municipal wastewater treatment (WWT) characterised by relatively high concentrations of nitrogen and phosphorus. Consequently, they do not adequately describe rate-limiting effects of nitrogen and phosphorus on microbial activity.

This master thesis summarises factors affecting nutrient requirements, problems caused by poor nutrient dosage strategies and some difficulties related to nutrient dosage control. An extended version of ASM1 for nutrient deficient conditions is proposed. Moreover, effects related to higher-order organisms within the biological system are considered. Measurements and experience from the WWT plant at Hyltebruk pulp and paper mill, owned by Stora Enso AB, are used as a starting point for the problem formulation, modelling work and parameter estimation. The work was carried out in co-operation with Anox AB.

It is established that knowledge about three magnitudes is indispensable when determining the nutrient requirements of a nitrogen and phosphorus deficient biological WWT process. They are: 1) The organic load, 2) The observed yield coefficient, i.e. the amount of produced sludge and 3) The nutrient contents of this sludge. The observed yield depends on factors such as the sludge retention time, the microfauna and the endogenous respiration rate. The nutrient contents depend, among other things, on the availability of exogenous ammonia nitrogen and soluble phosphorus.

In this work, the ASM1 is extended to include:

- a proposed wastewater fractionation for phosphorus including five phosphorus state variables;
- limiting effects of ammonia nitrogen and soluble phosphorus on heterotrophic growth;
- nutrient regeneration through predation of higher-order organisms (protozoa) on bacteria;
- variable phosphorus uptake by active biomass depending on the available amount of exogenous soluble phosphorus.

The specific two-stage design of the WWT plant presently in use at Hyltebruk, with production of dispersed bacteria in a biofilm (suspended carriers) stage, and subsequent consumption of these bacteria by protozoa in an activated sludge stage, provides great challenges with regard to nutrient modelling. During energy transfer from bacteria to protozoa, energy is lost (from the system) due to inefficient biomass conversion. The nitrogen and phosphorus parts of the heterotrophic biomass not used for synthesis of protozoan biomass will be regenerated to the water phase and eventually be available for growth of bacteria.

In the extended model, the time response of the system due to transient influent phosphorus concentrations, is affected by the variable phosphorus uptake of growing biomass. This phenomenon is in accordance with real data and explains why the propagation of dosage changes do not instantly reach the effluent. The presented model is validated for steady-state conditions; dynamic simulations demonstrate how the protozoa, the sludge retention time and the concentration-dependent phosphorus uptake process influence nutrient requirements and the performance of the treatment process.

The results of the measurement campaign indicate that the observed, total yield of the Hyltebruk WWTP is approximately 0.13 kg COD sludge produced/kg soluble COD reduced and that the mean nutrient content of the produced sludge averages 4% nitrogen and 0.4% phosphorus (on a COD basis). This relatively poor nutrient content is typical for pulp and paper wastewaters. It is concluded that the activated sludge stage of the process acts as a converter of particulate nutrients into soluble nutrients so that soluble nitrogen and phosphorus levels in the effluent theoretically cannot be zero.

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# 1 Introduction

Being one of the major consumers of natural resources (wood, water) and energy (fossil fuels, electricity), pulp and paper production leads to significant impacts on the environment. In terms of freshwater withdrawal, the pulp and paper industry ranks third in the world, after the primary metals and the chemical industries. The consumption varies with the type of paper being produced and can be as high as 60 m<sup>3</sup> per ton of paper produced (Thompson *et al.*, 2001).

If not efficiently treated, the effluent mill water will contain large amounts of organic matter and/or nutrients promoting microbial growth, eutrophication and oxygen consumption in the recipient. This may lead to impacts on the aquatic ecosystem, such as oxygen deficiency. The most frequent way to reduce organic matter in the effluents of pulp and paper mills is by biological oxidation processes. These are often coupled with mechanical treatment processes (e.g. sedimentation, flocculation etc) and with chemical treatment processes (e.g. phosphorus precipitation). When the wastewater is characterised, it is not possible to determine the exact concentrations of the different compounds in the water. Instead they are lumped together into broader categories such as total and dissolved chemical oxygen demand (COD), a parameter determining the amount of oxygen required to chemically oxidise the pollutants in a sample.

Hyltebruk Mill is a large producer of standard newsprint (850 000 tons/year), and has served the newspaper publishers and printers since 1972 when the first of the new paper machines was taken into operation. The mill is located by the Nissan River in a heavily forested part of Sweden's Småland province. In the fall of 2002, a new wastewater treatment plant (WWTP) was taken into operation at Hyltebruk. The basis for the plant design is a combined biofilm/activated sludge process (BAS), developed by Anox AB. The layout aims at providing robust operation with low sludge production and good sludge settleability.

As for most pulp and paper wastewater treatment (WWT) systems, phosphorus and nitrogen supplementation is required for the Hyltebruk WWTP as the wastewater does not contain sufficient nutrients for good biological treatment. A robust and efficient nutrient dosage control system would generate several benefits, such as reductions in nutrient discharges from the treatment system, fewer process disturbances (no disturbances due to low nutrient concentrations), decreased nutrient consumption and, consequently, lower costs. One of the main challenges in controlling the biological treatment is to add sufficient nitrogen and phosphorus to obtain high COD reduction without releasing the additives into the recipient, i.e. to ensure robust operation of the treatment system while minimising the effluent nutrient concentrations.

Prior to the development of a satisfactory nutrient dosage control strategy, knowledge about the biological response to conditions with low and transient nutrient concentrations must be obtained. Based on this information a mathematical model of the system is developed, which is important for control system design. Several mathematical models are available describing COD removal processes, for example the Activated Sludge Model No. 1 (ASM1) (Henze *et al.*, 2000). They have, however, primarily been adapted to municipal wastewater treatment characterised by relatively high concentrations of nitrogen and phosphorus. Consequently, they do not adequately describe rate-limiting effects of nitrogen and phosphorus on microbial activity.

## 1.1 Objectives

This work focuses on aerobic biological oxidation processes and, in particular, on how these processes can be modelled under conditions with low and transient wastewater nutrient concentrations. To achieve this primary objective, a number of secondary objectives can be stated: 1) Investigate the present knowledge of nutrient requirements for efficient treatment; 2) Identify observed problems with nutrient dosage control; 3) Adapt and condense the theories from 1) and 2) into a dynamic mathematical model; 4) Investigate the proportions of nutrient flows throughout the Hyltebruk WWTP and how they are influenced by the BAS process design; 5) Validate the developed model.

## 1.2 Main results

Essentially, nutrient requirements for a COD-reducing WWT system depend on the influent organic load, the amount of sludge produced (i.e. the observed yield) and the nutrient contents of this sludge. It is established that the sludge production and nutrient contents are affected by several factors, such as the microfauna, sludge retention time, endogenous respiration rate and the availability of exogenous ammonia nitrogen and soluble phosphorus.

A revised version of the ASM1 is presented. It has been extended to include:

- a proposed wastewater fractionation for phosphorus including five phosphorus state variables;
- limiting effects of ammonia nitrogen and soluble phosphorus on heterotrophic growth;
- nutrient regeneration through predation of higher-order organisms on bacteria;
- variable phosphorus uptake by active biomass depending on the available amount of exogenous soluble phosphorus.

The results of the measurement campaign indicate that the observed, total yield of the Hyltebruk WWTP is approximately 0.13 kg COD sludge produced/kg soluble COD reduced and that the mean nutrient content of the produced sludge averages 4% nitrogen and 0.4% phosphorus (on a COD basis). This relatively poor nutrient content is typical for pulp and paper wastewaters. It is concluded that the activated sludge stage of the process acts as a converter of particulate nutrients into soluble nutrients so that soluble nitrogen and phosphorus levels in the effluent theoretically cannot be zero.

## 1.3 Outline of the thesis

This thesis starts with a short description of pulp and paper manufacturing processes, the contents of the generated wastewater and some common ways to treat it. In the literature survey, mainly covered in Chapters 3 and 5, factors affecting nutrient needs and dosage control are condensed. To understand the qualities of the BAS design, some rather complex knowledge of biological oxidation processes must be gathered. Therefore, the Hyltebruk WWTP description is placed in Chapter 4, in-between the two theoretical chapters. In Chapter 6, some fundamentals of



activated sludge modelling are presented. The model development, the real intellectual challenge of the work, is presented concurrently. The chapter ends with a summary of the extensions that have been made to the original ASM1. In Chapter 7, the results of a detailed measurement campaign carried out at the Hyltebruk WWTP are analysed. These are used to fractionate the model input and for model validation. The thesis ends with investigations of model performance and, finally, conclusions. In the Appendix, detailed results from the calculations and analyses are found.

## 2 The processes

In this chapter, some of the key processes are described. First the processes producing the wastewater – the pulp and papermaking – are described. Then the main features of the wastewater treatment are presented.

### 2.1 Pulp and papermaking

Wood, one of the major raw materials used in the pulp and paper industry, is composed of cellulose fibres, carbohydrates such as starch and sugars, and lignin. In order to make the raw material suitable for papermaking, the wood is broken down to form a pulp. Three main processes are used to prepare wood pulp: In mechanical pulping, a block of wood, usually debarked, is passed through a rotating grindstone where the fibres are stripped off and suspended in water. Chemical pulping utilises chemicals to break down the wood in the presence of heat and pressure. The spent liquor is then either recycled or disposed of by incineration for heat recovery. Chemical thermo-mechanical pulping is a combination of the two. The wood is first partially softened by chemicals and the remainder of the pulping proceeds with mechanical force (Thompson *et al.*, 2001).

In the paper mill, pulp is converted into some type of paper product on the paper machine. Firstly, the pulp is diluted with whitewater (the process water in the paper mill) to such an extent that the dry substance ranges between 0.1-1%. Several different steps in the paper machine then dry the mixture to form paper.

### 2.2 Contents of pulp and paper wastewater

The mill wastewater contains of an abundance of different substances, both organic and inorganic. The particulate material is mainly made up of wood fibres that have not been retained on the wire in the paper machine. Soluble pollutants become part of the wastewater through different reject flows from purification and separation processes within the pulp and paper mill. They mainly originate from wood and recycled fibres that partly have dissolved during the manufacturing process and consist of lignin, carbohydrates, extractives and their degradation products. During the pulp processing some fibres are broken down into small fragments, which are referred to as fines. These fines can be further degraded so that short chains of polysaccharides are dissolved. The dissolved organic material from the wood pulping is split up between the pulp passing on to the paper machine and discharged wastewater (Alexandersson, 2003).

Different additives and chemicals from the mill may also be found as soluble components in the wastewater. The major contribution of additives in recycled pulp wastewaters is starch, which is added to increase the strength of the old product. In the papermaking process, fillers (a white clay) may be added to improve the product printability. In order to improve the retention of the

fillers, some chemicals, so called retention aids, are added to the whitewater (Alexandersson, 2003).

## 2.3 The Hyltebruk mill

At Hyltebruk, the virgin fibers used for papermaking originate from spruce and to some extent pine, wood mainly cleared of in southern Sweden. Recycled fibers from different wasted paper products (newspapers and magazines) are also used. Both virgin and recycled pulp is manufactured mechanically. In the ground wood pulp (GWP) process, logs of wood are pressed against a wet rotating grindstone. In the thermo mechanical pulp (TMP) process, wood chips are grinded against rotating steel plates. Huge electrical motors drive both processes. Recycled fibres are mixed mechanically with water and chemicals. A process called de-inking removes ink from this pulp. Mechanically manufactured pulp has a tendency for yellowing and therefore, hydrosulphite is added to the pulp.

The Hyltebruk paper mill has four paper machines producing 850 000 tons/year of newsprint. The most rapid machine has a velocity of 1500 meters per minute. Considering the dry substance content of the product, 95%, the process is very energy consuming. Each day, some 20 000 m<sup>3</sup> of wastewater is transported to the WWTP.

## 2.4 Basic wastewater treatment processes

Water quality requirements lead to general limitations on organic matter, nitrogen and phosphorus concentrations in the effluent from pulp and paper wastewater treatment plants. In Europe this policy is not only necessary due to eutrofication of lakes and slowly running waters, but also due to eutrofication and oxygen deficiency in the Baltic and North sea (Möbius, 1991). Normally, and also for the Hyltebruk WWTP, the main objective for treating pulp and paper wastewaters is to reduce large amounts of organic matter. The most common way to achieve this is to first remove solids and particles through mechanical treatment processes and then to reduce dissolved pollutants through aerobic carbon oxidation, a biological treatment process. To achieve high COD reductions by means of biological treatment, nutrients must normally be added to the nutrient deficient pulp and paper wastewaters. Still, the additives must not be released into the recipient. The nitrogen and phosphorus discharge from the pulp and paper industry may not account for a major part of a country's total discharge. However, the mills are often large integrated units and the local discharges of nutrients have more significant effects than the total figures suggest (Järvinen, 1997).

This work focuses on the biological processes and mainly on how these are carried out by activated sludge systems. Although also important, the mechanical processes are presented only in a short and simplified manner.

## 2.4.1 Mechanical treatment processes

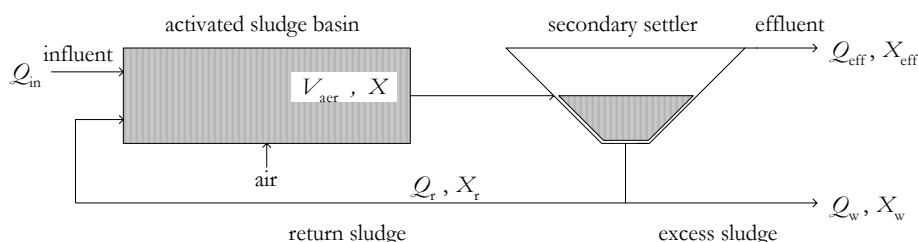
Mechanical treatment processes remove particulate matter from the wastewater. Sedimentation, or settling, is the most frequently used method to separate solid particles from the liquid phase. Particles that have a higher density than the surrounding liquid settle and accumulate at the bottom as sludge. Another common separation method is flotation where the density of the particulate material is lowered by means of air. The solids can then be removed from the top layer of the liquid. There are several ways to design mechanical treatment processes and many parameters influence the decisions. For example, the designers have to take into account the characteristics of the formed sludge, a factor influencing the sludge treatment processes and settling properties in subsequent settlers.

## 2.4.2 Aerobic biological wastewater treatment systems

Several plant designs for aerobic oxidation are available on the market. One of the most frequently used, the activated sludge system, is described below as well as the suspended carrier biofilm system. Both are presently in use at the Hyltebruk WWTP.

### Activated sludge systems

Historically, wastewater from the pulp and paper industries has been treated in huge aerated basins. Through aerobic biological oxidation processes, microorganisms in the basin convert the pollutant, mainly soluble COD, to particulate COD and carbon dioxide. The reaction products are separated as solids (through sedimentation) and gas. These processes are the essence of this work and the theories are presented in detail in Chapter 3. The treatment reaction rates are proportional to the amount of microorganisms present and therefore, as biomass grows relatively slow, the aerated basins require huge volumes for effective treatment. The activated sludge process solves this problem by recirculation of the biomass, see Figure 2.1. In this way, the sludge concentration in the aerated basin, or mixed liquor suspended solids concentration (MLSS), can be kept high thereby lowering the volumetric requirements.



**Figure 2.1.** Outline of an activated sludge system. Shaded areas symbolize sludge.

The activated sludge, which has given this type of process its name, is normally a thick brownish slurry that consists of the microorganisms and other particulate matter. It is mixed with the influent in the aerated activated sludge basin. Treated water with sludge is transferred to the settler where the sludge settles as sediment and cleaned effluent is withdrawn from the settler

surface. The majority of the sludge is brought back as return sludge and the surplus is wasted as excess sludge. Oxygen is transferred to the activated sludge basin by aerators. The aeration has two purposes, on the one hand it continuously stirs the mixture and on the other hand it provides oxygen to the microorganisms. Crucial for good separation performance is that the sludge settles well. Some aspects on microbial communities and their settling properties are discussed in Section 3.4.

The return sludge flow rate is set so that the MLSS supports good treatment properties and so that the sludge is distributed between the settler and aeration basin in an adequate way. A normal value in Sweden is to recirculate 50 to 100% times the influent flow rate:

$$\frac{Q_r}{Q_{in}} \in [0.5, 1] \quad (2.1)$$

where  $Q_{in}$  and  $Q_r$  denote influent and return sludge flow rates respectively. Another important design parameter is the aerobic hydraulic retention time ( $HRT_{aer}$ ). It is defined as:

$$HRT_{aer} = \frac{V_{aer}}{Q_{in}} \quad (2.2)$$

where  $V_{aer}$  (see Figure 2.1) is the aerobic volume.

### Suspended carrier biofilm systems

In a biofilm process, the microorganisms grow attached to a solid surface. Materials that are usually used as growth support are stones, sand or some type of plastic material. The process reactor is designed in such a way that the carrier material is kept in the reactor, e.g. through gravity, screens, etc. Thereby the sludge is kept within the system without recirculation. The process is thought to be less sensitive to hydraulic and organic load variations than a traditional activated sludge process.

### 3 Aerobic carbon oxidation

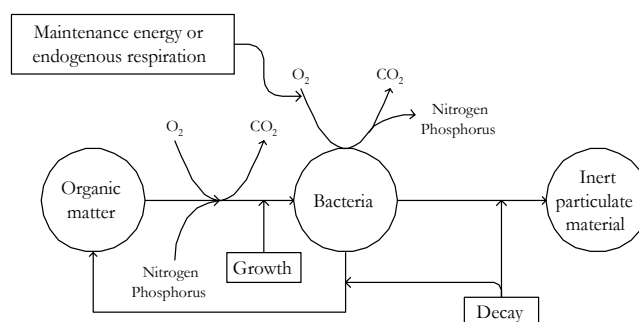
Basically, the removal of COD through biological oxidation is a separation process. During aerobic conditions, microorganisms oxidise organic matter to carbon dioxide for energy and assimilate it for cell synthesis. The carbon dioxide can then be separated from the system via the air while the produced sludge is separated as solids.

Of primary concern in the design and operation of industrial waste-treatment facilities are the rates at which the oxidation reactions occur, the amounts of nutrients required and the quantity (and quality) of the biological sludge they produce. The nitrogen and phosphorus content of the organic matter in pulp and paper wastewaters do not suffice for cell synthesis and as a result, these nutrients must be added to the wastewater. Ideally, nutrients present in an activated sludge system end up in the synthesized biomass. Thus, the amount of nutrients required for good treatment (including low levels of nutrients in the effluent) is closely linked to the influent organic load, the amount of biomass it yields and the nutrient content of this biomass.

Irrespective of which plant layout is used, two basic phenomena occur when organic matter is removed from solution by heterotrophic microorganisms (by definition, organic matter is the source of energy (substrate) for heterotrophs):

1. organic matter is oxidised (with an associated oxygen consumption) by the organisms for energy and
2. new cell mass is synthesised.

The oxidation process is illustrated in Figure 3.1 below: Firstly, energy is required for cell growth. Growing cells utilize exogenous substrate (located outside the cell membrane) as required for growth. A living cell needs energy permanently, regardless of whether it grows or not, because the biochemical reactions important to support essential activities cannot stop. Thus, after formation, the cell mass continues to oxidise organic matter for maintenance energy. This process is called endogenous (occurring inside the cell) respiration and involves utilization of stored/accumulated substrate and other cell materials not absolutely necessary for metabolic and growth functions. Finally, the cell loses its vital functions and lyses. Remaining cell materials are then available to other living cells in the culture.



**Figure 3.1.** An overview of the biological oxidation process.

A portion of the cell mass generated will remain as a nonbiodegradable (inert) fraction (Eckenfelder, 1992). The nitrogen and phosphorus contents of organic matter do not suffice for cell synthesis so these nutrients must be available in other forms. During the endogenous processes, nitrogen and phosphorus may be released. The substrate used for energy is separated as (gaseous) CO<sub>2</sub> to the air. In the activated sludge process, the part ending up as biomass and inert material is separated in the secondary settler unit.

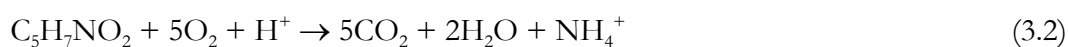
The amount of oxygen required to oxidise the organic content of a sample depends on concentration and type of organic in question. Two widely used methods of estimating the organic strength of a sample, COD and BOD (biochemical oxygen demand), are based on this corresponding oxygen demand. These are described below together with a more detailed description of the oxidation process.

### 3.1 Estimating the organic content

The analysis of aerobic biological oxidation requires that the organic matter can be described using chemical formulas. Wastewater contains thousands of different organics and a measurement of each component would be impossible. Instead, the organic matter can be symbolized by the average composition C<sub>18</sub>H<sub>19</sub>O<sub>9</sub>N (Henze *et al.*, 1995). The (balanced) reaction expression for aerobic oxidation of substrate reads:



COD measures the amount of oxygen required for chemical oxidation of organic matter in a sample. It is stated in the unit oxygen, for example mg O<sub>2</sub>/l. In Reaction 3.1, 17.5 mol O<sub>2</sub> is used to oxidise 1 mol substrate. The COD-value of substrate becomes (using molar masses) 17.5·2·16 g O<sub>2</sub>/(18·12+19+16·9+14) g substrate=1.42 g O<sub>2</sub>/g substrate. Like for substrate, the synthesized cell mass can be described with an average chemical composition: C<sub>5</sub>H<sub>7</sub>NO<sub>2</sub> (note that phosphorus, due to its low cell content, is not included in the average compositions given above). Aerobic oxidation of biomass then occurs following:



The COD of the process is calculated in the same way as for substrate, which gives 1.42 g O<sub>2</sub>/g biomass. Thus, the conversion factors of the assumed compositions of biomass and organic matter in both cases are 1.42 g COD/g matter.

Although the COD analysis can be carried out automatically and relatively fast (1-2 hours), it does not give a relevant picture of the pollutant content from the microorganisms point of view. Chemical oxidation usually eliminates all organic matter in a short time span. In reality, the oxidation of organic matter by activated sludge is a slow process, which may require several days. Also, part of the substrate will be inert to the organisms, no matter how much time they are given. In order to design, or fairly diagnose the performance of a biological process, information about the pollutants biodegradability and at which rate it can be degraded is needed. Therefore, the COD tests often are complemented with the more time consuming BOD test.

BOD is the quantity of oxygen utilised by a population of microorganisms in the aerobic oxidation of organic matter in a sample of wastewater at a temperature of 20 °C. The unit of BOD is the same as for COD, mg O<sub>2</sub>/l. In contrast to COD, the BOD is time dependent and can be described as:

$$\text{BOD}(t) = S_{\text{O}}(0) - S_{\text{O}}(t) \quad (3.3)$$

where  $S_{\text{O}}$  is the dissolved oxygen concentration at time 0 and time  $t$ . Normally, BOD analyses are carried out during five or seven days. During this time, typically two-thirds of the biodegradable organic matter is oxidised. By extending the duration of the BOD<sub>5</sub> analysis to somewhere between 20 and 30 days a more complete analysis is obtained. The oxygen consumption is measured as a function of time and after extrapolation of these data, the ultimate BOD, BOD<sub>∞</sub>, is obtained. The COD value of the remainder will not be zero but a measure of the amount of organic matter not available for microbial oxidation.

A third way to estimate the organic content of a sample is TOC (Total Organic Carbon). TOC measures the total organic carbon in the sample. It can be related to COD through a carbon-oxygen balance. Reaction 3.1, for example gives a COD:TOC ratio  $17.5 \cdot M_{\text{O}_2} / 18 \cdot M_{\text{C}} = 2.6$  for substrate. Depending on the organic in question, the COD:TOC ratio may vary from zero when the organic material is resistant to chemical oxidation to 5.33 for methane (Eckenfelder, 1989). A common COD:TOC ratio for wastewater is around 3.

Throughout the treatment plant, biodegradable organic matter will decrease. This will be seen in the COD, BOD and TOC analyses. The inert organic matter, however, is only seen in the COD and TOC measurements. Thus, a decrease in BOD:COD and BOD:TOC ratios throughout the plant is expected.

From a control perspective, BOD is not a suitable measurement variable as the results are delayed with at least five days from the sample occasion. Instead, control based on the organic load depends on empirical correlations between COD and BOD on the one hand or TOC and BOD on the other. If the influent organic load does not show significant variations in its constitution, such a correlation is possible to find. However, attention should be paid to the variations that are likely to occur, especially during upstream in-mill process changes and closedowns.

As the nutrients added to an activated sludge system ideally ends up in synthesized biomass, the amount of nutrients required for good treatment (including low levels of nutrients in the effluent) is closely linked with the influent BOD and the fraction of it that is converted to biomass. This fraction is given by the carbonaceous yield constant.

## 3.2 Carbonaceous yield constant

The extent to which reactants are converted to products is expressed as the reaction yield. In aerobic oxidation processes, the amount of substrate generating cell mass is of great importance:





where  $Y$  is called the true or theoretical carbonaceous yield constant. Generally speaking, yield is the amount of product formed or accumulated per amount of reactant provided or consumed. There is no strict definition of yield; several different yield parameters are applicable in different situations. The yield is often expressed in the units kg biomass/kg organic or kg biomass COD/kg substrate COD. In connection with aerobic growth, microorganisms have an energy efficiency of 55-60%. The maximum yield constant measured in COD units is therefore 0.55-0.6 (Henze *et al.*, 1995).

The observed yield does not equal the theoretical yield. When considering biological processes we are always lumping different individual reactions together. If, for example, the total mass of substrate consumed is  $\Delta S$ , some portion  $\Delta S_1$  is used for biomass growth ( $\Delta X$ ) while the remainder,  $\Delta S_2$ , is channelled into other products and biological activities not related to growth. The observed biomass yield is:

$$Y^{\text{obs}} = \frac{\Delta X}{\Delta S_1 + \Delta S_2} \quad (3.5)$$

In comparison, the true or theoretical yield from substrate is:

$$Y = \frac{\Delta X}{\Delta S_1} \quad (3.6)$$

(Doran, 1995). True yields are stoichiometric coefficients based on metabolic pathways. Normally these are complex and difficult to calculate. However, theoretical yields can be related to observed yields. This is illustrated in the simplified example below where the bacterial population  $X$  grow on substrate  $S$  in a batch culture:

$$\frac{dX}{dt} = (\mu - b)X \quad (3.7)$$

$$\frac{dS}{dt} = -\frac{\mu}{Y}X \quad (3.8)$$

where  $\mu$  = growth rate,  $b$  = decay and endogenous respiration rate and  $Y$  = true or theoretical biomass yield coefficient (mass  $X$  formed/mass  $S$  consumed). Combining Equations 3.7 and 3.8 gives the observed yield coefficient:

$$Y^{\text{obs}} = \frac{dX}{dS} = Y \frac{(\mu - b)}{\mu} \quad (3.9)$$

The observed yield is, due to endogenous processes, lower than the theoretical. As we will see later, the observed yield over the Hyltebruk activated sludge reactor is negative.

The majority of the nutrients added to an activated sludge treatment plant are removed from the system through the excess sludge. The amount of nutrients that assimilate in new biomass is therefore of crucial importance for establishing nutrient requirements.

### 3.3 Nutrient contents of activated sludge

Nutrients are by definition chemical components utilized for cell synthesis and they can be divided into two groups: Macronutrients are those found in biomass in larger fractions (such as C, O, H, N, P, S, Ca, K, Mg) while micronutrients (such as Fe, Mo, Zn, Co) are found in small fractions. In practice, and in this work, the term nutrient is used for nitrogen and phosphorus only as these are the limiting nutrients in pulp and paper wastewater treatment.

Phosphorus is an essential element for bacterial growth. Phosphorus is used for energy yielding reactions in ATP and NADPH, it is also used in DNA and RNA genetic material as well as in structural components such as phospholipids. The main function of nitrogen, also essential for growth, is as a constituent of proteins (Richard, 1999).

Typical municipal wastewater activated sludge contains 1-3% phosphorus on a dry weight basis. In the average compositions of organic matter and biomass given above, phosphorus due to its relatively low content is not included. A typical bacterial cell can, however, be assumed to have the following chemical composition:  $C_{106}H_{181}O_{45}N_{16}P$ , which gives a stoichiometric P content of 1.3%. Pulp and paper activated sludges have a phosphorus content ranging from 0.5 to 1.5% by weight (Richard, 1999). The nitrogen content of biomass in municipal wastewater ranges from 8-12% (Henze *et al.*, 1995). Also for nitrogen, pulp and paper activated sludges seem to have lower contents. Saunamäki (1994) found that the nitrogen content in dry biomass was in the range 2.5-4.5%.

The generally recommended nutrient supply to activated sludge are one kg of phosphorus and five kg of nitrogen for every 100 kg of BOD oxidised (a BOD:N:P ratio of 100:5:1). The exact amounts of required nitrogen and phosphorus are dependent on the observed carbonaceous yield factor, which is a variable depending on the substrate type. Assuming an observed yield of 0.5 kg of bacteria produced per kg of COD oxidised, the stoichiometric need for nitrogen and phosphorus would be a BOD:N:P ratio of 100:4.6:0.65 based on the chemical formula stated above (Richard, 1999). Many pulp and paper wastewater systems require less than the traditional BOD:N:P (100:5:1) ratio. According to Möbius (1991), in-plant optimisation of nutrients in the influent leads to about 3.5 parts of N and 0.6 parts of P for 100 parts of BOD. Saunamäki (1994) found that an optimum BOD:P ratio was 100:0.4 resulting in less than 0.3 mg/l soluble phosphorus in the effluent (<0.1 mg  $PO_4$ -P/l).

### 3.4 Influence of microfauna on carbon oxidation processes

Generally, large amounts of bio-sludge are formed in aerobic biological wastewater treatment processes and the separation, dewatering, treatment and disposal of this sludge represent major investment and operating costs. The microfauna in a carbon oxidation system, i.e. the

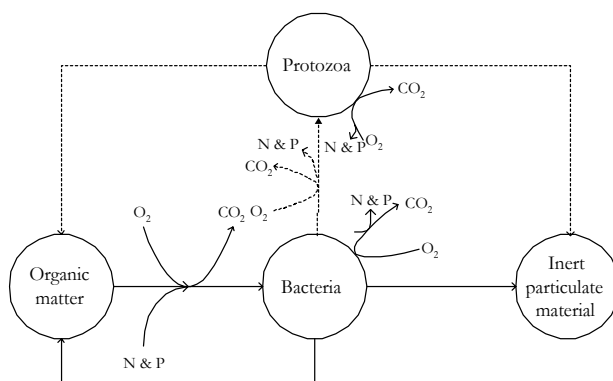
participating microorganisms, influence sludge characteristics in two ways. Firstly, it affects the settling properties of the sludge and secondly, it affects the observed yield of the WWTP.

In an activated sludge system, the particulate material is made up of several types of microorganisms and to describe them all fall beyond the scope of this work. However, the most important species and their characteristics must be mentioned in order to adequately describe the Hyltebruk WWTP system. In this work, microbial description is reduced to involve bacteria and protozoa/metazoa. Only heterotrophic bacteria (which use soluble organic matter as substrate) are discussed and three types of them are mentioned: dispersed bacteria, floc-forming bacteria and filamentous bacteria. The protozoa/metazoa are higher-order organisms, which graze on the bacteria. Henceforth, protozoa will be used as a lumped name describing all types of higher-order organisms.

It is imperative when treating wastewaters (industrial and municipal) by an activated sludge process that the produced sludge flocculates and settles well. The three types of bacteria and protozoa mentioned above are all present in an activated sludge plant and the maintenance of good settling properties requires balanced distribution of the microbial species and therefore involves engineered operational control. Examples of manipulated variables that influence the distribution of different forms of bacteria are the food to microorganism (F/M) ratio, oxygen concentration, and temperature. Some aspects on the control of filamentous bulking are discussed in Section 5.1. Dispersed or free-swimming bacteria are, as the name suggests, present in the wastewater clear phase. These bacteria do not settle and as a result, selection will prevent the dispersed bacteria population to grow large in an activated sludge process. In contrast, selection favours floc-forming bacteria, a species having the ability to produce a slime layer around them, which will glue them together into large particles with a high settling velocity, so called flocs. Ideally, a small fraction of filamentous bacteria (bacteria growing in string-like formations) bringing together some flocs into bigger flocs are present as well. In contrast, an excess-growth of filaments will deteriorate sludge settling properties dramatically.

Not only the washout of dispersed bacteria favour growth of floc-formers. Also, grazing by protozoa suppresses the growth of dispersed bacteria and favours floc or film forming bacteria as these are more protected against predation. Protozoa also contribute to the cleansing of the effluent water as they eat or filter the dispersed non-settling bacteria.

One way to reduce high sludge production is to exploit the organisms in the activated sludge process that predate on the (sludge) bacteria. If such higher organisms, here named protozoa, are present in significant concentrations, an extra growth step is realized, which converts the sludge into new biomass, water and CO<sub>2</sub>. This is illustrated in Figure 3.2. During energy transfer from bacteria to protozoa, energy is lost due to inefficient biomass conversion (Ratsak *et al.*, 1995). This amounts to a decreased biomass formation, that is, lower sludge production.



**Figure 3.2.** The influence of protozoa on carbon oxidation and sludge production.

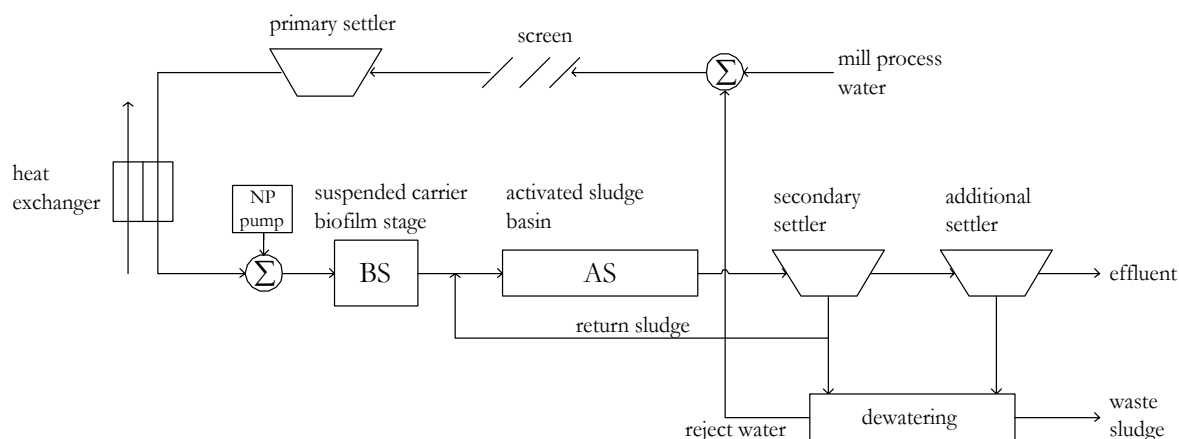
## 4 Hyltebruk wastewater treatment plant description

Based on the selection pressures together with the fact that the net energy efficiency (observed yield) decreases with an increased number of ecological levels (Section 3.4), Anox AB has developed a combined biofilm/activated sludge process layout (BAS), which aims at providing a robust WWT configuration with low sludge production and good sludge settleability. This process principle was the basis for the design of the present WWTP at Hyltebruk.

The idea is to first treat the wastewater in a high-rate biofilm process, thereby removing the readily biodegradable organic matter. The remaining COD is then reduced in a traditional activated sludge process. In the biofilm reactors, bacteria grow attached to a carrier material and form a biofilm. A short hydraulic retention time suppresses the growth of protozoa and floc-formers in the water phase and favour the growth of dispersed bacteria. In the subsequent AS process, the dispersed bacteria are consumed by protozoa in a predator-prey relationship.

When working with (or reading about) a WWT system, it is of great importance to get a feeling for the plant and its performance. Therefore, a simple and concise plant description is given below. The plant characteristics given are by no means originating from advanced experiments and should not be considered as results of this work. Instead, they represent mean values obtained from the Hyltebruk water laboratory during the spring and summer of 2003 and from experience of the operators.

In the fall of 2002, the new Hyltebruk WWTP was taken into operation. The main stages that form the system are a primary sedimentation stage, a suspended carrier biofilm stage, an activated sludge stage, a secondary sedimentation stage and finally an additional sedimentation stage. The last stage allows for occasional chemical phosphorus precipitation. The plant is constructed with two identical lines in parallel (line 1 and line 2) and each stage in the overall WWTP therefore consists of two units: there are two biofilm reactors, two activated sludge reactors, two secondary settlers and two additional settlers. A schematic layout of the system is given in Figure 4.1. The WWTP treats approximately 20 000 m<sup>3</sup>/d of wastewater. In the first two treatment stages, the screen and the primary settler, a substantial part of the particulate COD is removed. Scrapes transfer bottom sludge from the primary settler to the outlet point. The waste sludge (containing mainly fibers) is then dewatered in the sludge treatment process. Before entering the two biological stages, on which this work focuses, the warm mill effluent is heat-exchanged to reach a temperature of 35 °C. The flow is then split into the two lines. Nutrients in the form of urea and NP granules are then added to both flows.



**Figure 4.1.** Flow chart of the Hyltebruk wastewater treatment plant. Each stage from the suspended carrier biofilm stage to the additional settler consists of two units in parallel.

## 4.1 The suspended carrier biofilm stage

The two biofilm reactors are designed as continuously stirred tank reactors (CSTRs). Each has a volume of  $2500 \text{ m}^3$  and the hydraulic retention time (HRT) is 6 hours. The reactors are filled (fill factor 40%) with carriers of type NATRIX optima®, a cylindrical plastic carrier with a height of 50 mm and a diameter of 60 mm. The carriers have been chosen so that they support good oxygen transfer, decrease the risk of clogging and facilitate mixing. A screen forces the carriers to remain within the reactor. Wastewater entering the biofilm stage (BS) has a soluble COD concentration averaging  $2200 \text{ mg COD/l}$ , which result in a total (both lines) soluble organic load of  $44\,000 \text{ kg/d}$ . A high biomass concentration, typical for suspended carrier systems, yields a high COD reduction rate and capacity. The short HRT favours growth of dispersed free-swimming bacteria. Approximately 45% of the influent soluble COD is reduced in this stage; the BS effluent, thus, has a soluble COD concentration of  $1200 \text{ mg/l}$ . The main properties of the BS are given in Table 4.1.

Volume	$\text{m}^3$	$2 \times 2\,500$
Influent flow rate	$\text{m}^3/\text{d}$	20 000
HRT	h	6
COD reduction	%	45

**Table 4.1.** Some main properties of the biofilm stage.

The sludge in the BS consists of particulate material not removed in the primary sedimentation stage, material that has been stripped from the carriers and active biomass. The two latter fractions are products from biological oxidation. Roughly,  $400 \text{ mg SS/l}$  or  $500 \text{ mg COD/l}$  are produced in the BS.

## 4.2 The activated sludge stage

The activated sludge basins are designed as plug-flow reactors encircling the biofilm basins. Their volumes are 7500 m<sup>3</sup> each giving an HRT of 18 hours. As for the BS, the knowledge of sludge production (and consequently the sludge age) is rather restricted. However, the sludge age is in the magnitude of 9 days. This supports growth of protozoa and floc-forming bacteria, and due to these degraders, the soluble COD remaining from the biofilm treatment is reduced almost immediately. About 300 mg soluble COD/l remains in the WWTP effluent as an inert component. Considering the total influent organic load, the reduction of soluble COD in the activated sludge stage (AS) is approximately 42%. The AS main properties are given in Table 4.2

Volume	m <sup>3</sup>	2×7 500
Influent flow rate	m <sup>3</sup> /d	20 000
HRT	h	18
Return sludge flow rate	m <sup>3</sup> /d	20 000
Effluent flow rate	m <sup>3</sup> /d	19 000
COD reduction	%	41
Sludge age	d	9

**Table 4.2.** Some main properties of the activated sludge stage.

The sludge produced in the primary settler, secondary settlers and in the additional settlers is dewatered in the sludge treatment stage. The supernatant is recirculated as reject water to the WWTP influent while the sludge is incinerated.

The treated effluent water discharge is approximately 6000 kg COD/d. The phosphorus and nitrogen loads are more varying and uncertain. Although not finally decided, the local government allows Hyltebruk to discharge 17 kg P and 150 kg N per day on a yearly basis.

## 5 Nutrient requirements and dosage control

As for most pulp and paper wastewaters, phosphorus and nitrogen supplementation is for the Hyltebruk WWTP as the wastewater lacks sufficient nutrients for appropriate biological treatment. Wastewater treatment at insufficient nutrient levels often results in poor sludge quality with a sludge that settles and compacts poorly, may foam, and dewater poorly (Richard, 1999). As a large part of the nutrients are incorporated in particles, an increase in effluent solids concentration (caused by poor sludge quality) cause increased levels of phosphorus and nitrogen in the effluent. Severe nutrient deficiency may also affect the main purpose of the treatment plant, that is, reducing the influent COD.

Optimisation of nutrient dosage aims at minimising dosage (costs) without jeopardising the operating stability of the biological process (Möbius, 1991). However, especially the fear of sludge bulking and subsequent discharge of suspended solids makes it difficult to realise the idea of lowering nutrient dosages (Jansson *et al.*, 1994).

Part of the material in Sections 5.1 and 5.2 is based on Richard (1999).

### 5.1 Problems caused by poor nutrient dosage strategies

The excess growth of filamentous organisms is well known to cause problems for the operation of activated sludge plants. Filamentous organisms generate sludge that settles poorly resulting in higher TSS (total suspended solids) levels in the effluent. Although sludge bulking is caused by several factors, including low oxygen concentration, septicity (high organic acid concentration), low F/M (food to microorganism ratio), low pH and low nutrient concentrations, its prevention demands sufficient nutrient levels. It is believed that any growth-limiting conditions favour filaments as these organisms have a higher growth rate during conditions of low concentrations of substrate, oxygen or nutrients. In addition, the maintenance energy requirements of filamentous microorganisms are usually lower in comparison with non-filamentous microorganisms. Therefore, the former microbes have stronger resistance to starvation than the latter ones (Eckenfelder *et al.*, 1992).

A significant problem in pulp and paper wastewater treatment systems (and other industrial wastewater systems) is the overproduction of exocellular polysaccharide at phosphorus and nitrogen deficiency. Treatment bacteria at nutrient deficiency continue to oxidise COD, but cannot synthesize “complete” bacteria due to the lack of nutrients. The partially oxidised COD intermediates are shunted out of the bacterial cell as exocellular polysaccharide to avoid internal carbon overload. This condition is manifested by an accumulation of gelatinous polysaccharide around the bacterial cell and higher effluent COD and TSS concentrations (Richard, 1999).

Moderate nutrient deficiency results in the sludge quality problems stated above. Severe deficiency results in insufficient nutrients to supply the energy-yielding reactions of the waste treatment bacteria. This results in incomplete soluble COD removal and an increase in the effluent soluble COD (Richard, 1999).



## 5.2 Factors affecting nutrient requirements

There is a great variation in the beliefs in the lower limit of nutrient dosage to activated sludge plants. A very common opinion is that there is a certain limit for residual concentration of phosphorus and especially nitrogen, below which the sludge starts to change and suffer with the result of poor reduction of organics and sludge characteristics (Jansson *et al.*, 1994). Wastewater treatment bacteria take up biodegradable substrates by facilitated (enzyme) transport, while inorganic nutrients such as nitrogen and phosphorus must enter the cells by diffusion. The facilitated carbon uptake rate can exceed the phosphorus diffusion rate, causing phosphorus deficiency within the biomass. This imbalance can only be overcome by increasing the phosphorus concentration around the biomass, causing an increase in the phosphorus diffusion rate into the biomass (Richard, 1999). As a consequence of this, rapidly assimilated organic matter require higher exogenous nutrient concentrations than do more slowly degradable carbonaceous substrates.

Due to the nutrient diffusion limitations described above, the wastewater treatment system configuration affects the amount of nutrients needed. Systems with a plug-flow configuration and thus a high organic load at the head-end of the basin may need higher nutrient levels due to the bacterial activity present and diffusion limitations. The same problem can occur in systems with selectors ahead of the aeration basin(s), due to high soluble BOD uptake rate in the selector. Lower organically loaded systems and those with completely mixed conditions have a lower BOD removal rate, and usually perform well at lower phosphorus concentrations (Richard, 1999).

It is difficult to quantitatively show the required nutrient residual concentration. While the effluent from a nutrient deficient activated sludge plant contains mostly non-available nutrients, only the biological available nutrients influence the diffusion rates. Often, there is lack of knowledge about these various types of nutrients in a plant and, as a result, it might be difficult to judge whether or not the correct amount of nutrients are dosed based only on the effluent total P and total N measurements. Also, if the effluent water is analysed as total (not filtrated) samples, the nutrients associated with particles must be subtracted from the total value to achieve knowledge about excess nutrients. However, even if this is carried out, the problems with distinguishing soluble biodegradable nutrients from soluble non-biodegradable nutrients still remains.

The sludge age is an important parameter in activated sludge processes. The aerobic sludge age, or aerobic sludge retention time  $SRT_{acr}$ , is the mean cell residence time of sludge (biomass) in the aerated volume:

$$SRT_{acr} = \frac{V_{acr} \cdot X}{Q_{eff} \cdot X_{eff} + Q_w \cdot X_w} \quad (5.1)$$

where  $Q_{eff}$  and  $Q_w$  denote effluent and wastage sludge flow rates respectively and  $X$  denotes biomass concentration. See also Figure 2.1. Cell decay releases nutrients that are internally recycled, reducing the required nutrient addition for good treatment. At longer sludge ages, more phosphorus is recycled due to the endogenous process and less phosphorus is needed per kg of BOD treated. Some longer sludge age systems have been successful at a BOD:P ratio of 100:0.4

due to a high cell endogenous rate. Conversely, lower sludge age activated sludge systems usually require close to the stoichiometric phosphorus requirement of 100:1.0 (Richard, 1999). The cell endogenous rate is also linked to the wastewater treatment temperature. Bacterial use of BOD for cell maintenance does not require exogenous phosphorus and may actually release it. This was illustrated in Figure 3.1. Cell maintenance is strongly temperature dependent and is lower at lower temperatures. At lower temperature, more of the BOD is available for cell growth and nutrient needs per kg of BOD are higher at this condition. Cell maintenance needs are higher at higher temperature, and this leaves less BOD for cell growth and lower nutrient needs per kg BOD treated.

In several contexts above, references have been made to the ‘BOD:N:P ratio required for good treatment’. The expression is seen frequently in the literature, but its’ meaning is not straightforward. In the expression, BOD should be understood as the total amount of organic matter that is to be removed from the influent wastewater. The ‘rule of thumb’ ratio, 100:5:1, does not imply that the N and P content of formed biomass is 5% and 1%. With an observed yield of 0.5, 100 kg BOD yields 50 kg sludge, and if the ‘rule of thumb’ ratio has been dosed in the influent, this sludge (hopefully) contains 2% P and 10% N. The point is, that nutrient requirements are closely linked to the amount of biologically produced biomass.

Different ways to calculate sludge production ( $SP$ ) are needed, depending on the purpose of application. As sludge treatment and sludge disposition often represent significant costs for a treatment plant, the total amount and not the source of sludge, often is in the focus. Therefore, the sludge production of an activated sludge plant traditionally is defined as the amount of particulate material removed from a WWTP per day. Furthermore, a common simplification is to neglect suspended solids present in the effluent so that the sludge production equals the excess sludge outtake. A consequence of this definition ( $SP_{\text{trad}}$ ) is that, if the suspended solids concentration is high in the WWTP influent, the  $SP$  can be large even with modest biological sludge production:

$$SP_{\text{trad}} = Q_{\text{eff}} X_{\text{eff}} + Q_w X_w \quad (5.2)$$

Nutrients are dosed to nutrient deficient bioprocesses in order to be assimilated into new biomass. Therefore, this work is focused on biologically produced sludge and influent sludge is deducted from the traditional  $SP$  expression:

$$SP = Q_{\text{eff}} X_{\text{eff}} + Q_w X_w - Q_{\text{in}} X_{\text{in}} \quad (5.3)$$

where  $SP$  is the definition of sludge production used throughout this work. A consequence of this definition is that it allows negative values of sludge production. This might be the case if concentrations of particulate matter in the influent are high.

## 5.3 Controlling nutrient dosage

The main control objective of an activated sludge plant, treating pulp and paper wastewater, is to keep the COD concentrations in the effluent at an acceptable level. This must be achieved without releasing a surplus of nutrients into the recipient. A well working control system generates several benefits:

- reductions in nutrient discharges from the treatment system;
- fewer process disturbances (no disturbances due to low nutrient concentrations) and
- decreased nutrient consumption and, consequently, lower costs.

In this section, the present nutrient control strategy at Hyltebruk is mentioned. A short summary of instruments available on the market is then given. Finally, some examples on how a control system could be designed are discussed. The control strategies presented are the result of a literature survey and in this work - they have not been evaluated in practice.

### 5.3.1 Present control strategy

To operate the Hyltebruk biological process, phosphorus and nitrogen must be added to the nutrient deficient wastewater. Nitrogen is added in the form of urea, a common fertilizer with the chemical formula  $(\text{NH}_2)_2\text{CO}$ . Urea dissolves slowly when diluted in water and ends up as ammonium and  $\text{CO}_2$ . Phosphorus is added in the form of NP granules. This salt consists of  $\text{NH}_4\text{-N}$  (14%),  $\text{NO}_3\text{-N}$  (12%) and  $\text{PO}_4\text{-P}$  (6%). The nitrate is not taken up as fast as the ammonium by the biomass. The motive for dosing nitrate is that part of the total nitrogen may be kept available until the activated sludge process.

The nutrients enter the WWTP just after the splitting of the wastewater into the two lines. They are dosed from two silos from where they are transported (diluted in water) to the addition point. Measurements made by operating personnel at Hyltebruk have shown that the nutrients are well split into the two lines. The nutrient dosage today is based on the hydraulic inflow, i.e. it is flow rate controlled. The flow rate, together with a set point, tells how much component (Urea or NP) that should be added to a certain amount of wastewater, for example  $0.02 \text{ kg NP/m}^3$ . Today, the set point is not controlled automatically. Instead, the operators change it manually based on changes in phosphorus concentrations in the effluent and treatment efficiency.

The full Hyltebruk WWTP design includes continuous TOC and phosphorus sensors. The intention is to use these for feedforward and feedback control (see below) but due to practical problems, this work has not yet been initialised.

### 5.3.2 Possible control strategies

#### Instrumentation

The control strategies discussed in this work require a number of on-line sensors. Sensors measuring pH, dissolved oxygen and flow rates are well established and generally available in many treatment plants. Others are at the forefront of new commercial sensor applications. These sensors include TOC, COD, phosphate and ammonium sensors. Sensors are available from a number of suppliers and are based on different measuring mechanisms. A survey (Alexandersson, 2003) shows that most of the ammonium sensors work in the concentration range of 0-20 mg  $\text{NH}_4\text{-N/l}$  and with a lower detection limit of approximately 0.05-0.1  $\text{NH}_4\text{-N/l}$ . Phosphate sensors generally measure in a range of 0-10 mg  $\text{PO}_4\text{-P/l}$  with a lower detection limit of 0.01-0.05 mg  $\text{PO}_4\text{-P/l}$ . Both ammonium and phosphorus sensors are afflicted with response delays of 5-15 minutes. TOC and COD on-line sensors can be used to estimate the COD concentration in the water stream. The measurement range of these sensors is generally 0-20000 mg/l (Alexandersson *et al.*, 2003a).

#### Feedback control

The most straightforward control approach is to measure nutrient levels in the effluent. The nutrient feed is then varied so that a desirable effluent concentration is achieved. To avoid nutrient deficiency within the biomass, a certain amount of ammonia and phosphorus must be found in the effluent. Typical guidelines for  $\text{NH}_4\text{-N}$  and  $\text{PO}_4\text{-P}$  in the effluent are 2-3 mg N/l and 0.05-0.1 mg P/l, respectively (Hynninen *et al.*, 1995).

It is well known that long time constants, i.e. the time it takes for a change to propagate through and affect the system, make the control of a system more challenging. In biological processes, most of the time constants associated with the biological activity are in the range of hours to days and even weeks. The biomass in an activated sludge system needs at least one sludge age to be replaced and consequently, steady-state conditions are unlikely to be found until at least 10 days after a change in nutrient dosage. Examples of these long time constants are given in Richard (1999) where the reduction of filaments following a phosphorus dosage increase took three to four sludge ages. Richard (1999) also found that in practice, it is very difficult to lower a residual phosphorus concentration even by closing the dosage of phosphoric acid completely. The biomass nutrient uptake rate varies and variations in effluent nutrient concentrations can have other reasons than proportional variations in the dosage. Saunamäki (1994) showed that a sudden phosphorus increase in the dosage feed raised the P content of biomass from 0.5% to 0.8%. High nutrient dosages like this without an increase in effluent nutrient concentrations has sometimes helped to stabilize the sludge volume index (sludge quality) (Möbius, 1991). A lowering of the nutrient dosage could, at least theoretically, deteriorate the sludge quality and the COD reduction and, hence, increase nutrient levels in the effluent.

Due to the long time constants and uncertain system responses exemplified above, a regular P controller or manual feedback control involves the risk of instability and, as an implication of this, the controller gain has to be set relatively moderately resulting in a slow controller. An

advantage with the time delays is that the system cannot be considered time critical. There is sufficient time to measure (with the delays related to actual sensor techniques), to process data and calculate control actions.

Another problem is where to measure the output. Even if the phosphorus level at the end of the activated sludge reactor is close to zero, significant levels may be found in the sedimentation unit overflow. This can be due to the ability of phosphorus accumulating organisms to release phosphorus in anoxic or anaerobic zones of the settler. Also, if the set points are set very low (typically < 0.1 mg/l for soluble phosphorus) on-line measurements might be unreliable.

### Nutrient-balance method

A drawback of a simple feedback strategy is, as discussed above, that nutrients, especially phosphorus, can be stored in excess in the cells and rapidly be released during certain conditions. Increased dissolved nutrients in the treated effluent can also be caused by oxygen deficiency in aerated zones, abnormal pH, or influent containing mineral oil or other substances toxic to microorganisms (Hynninen *et al.*, 1995). Thus, over- or under-dosage is not always shown in residual nutrient concentrations and a simple feedback control strategy must, with respect to these problems, be connected with biomass analysis. The goal would be to minimize changes in nutrient levels in the activated sludge biomass while monitoring and controlling dissolved nutrient levels in the effluent.

One direct way of calculating the required amount of nutrients is through mass balances. For dosing purposes the nutrient balance is determined so that the nutrient content in the influent (including the dosed nutrients) equals the nutrient discharge. For a simple activated sludge system with inflow *in*, effluent *eff*, and wasted sludge *w*, the steady-state mass balances will read:

$$Q_{in}(S_{nTOT,in} + X_{nTOT,in}) + G_{n,dos} = Q_{eff}(S_{nTOT,eff} + X_{nTOT,eff}) + Q_w(S_{nTOT,w} + X_{nTOT,w}) \quad (5.4)$$

where *n*TOT refers to the total concentration of nutrient *n* (N or P), *G* is the mass of nutrient dosed per time unit and *S* and *X* denote soluble and particulate concentrations, respectively. Normally, the mass flow of nutrients from the upstream process is not a manipulated variable, so this value is fixed. Regarding the effluent, it is vital to adjust the dosage so that nutrient levels are sufficiently high to generate good treatment efficiency. There is probably a certain limit (the set point) for residual concentrations. If the plant functions well, the particulate fractions of total N and P in the effluent should be low. Hence, with a desired wastage flow rate and fairly constant values of nutrient content in the sludge, the required amount of nutrients, *G<sub>n,dos</sub>*, can be calculated. The use of mass balances has inspired the nitrogen and phosphorus requirements equations presented in Eckenfelder (1989) (also cited in Grau (1991)):

$$N = 0.123 \frac{X_d}{0.8} \Delta X_v + 0.07 \frac{0.8 - X_d}{0.8} \Delta X_v \quad (5.5)$$

$$P = 0.026 \frac{X_d}{0.8} \Delta X_v + 0.01 \frac{0.8 - X_d}{0.8} \Delta X_v \quad (5.6)$$

Here, N and P represent required mass flows of total bio-available nitrogen and phosphorus (kg/d).  $\Delta X_v$  symbolize waste sludge flow-rate and  $X_d$  is the degradable fraction of the sludge: It is assumed that the biodegradable and non-biodegradable fractions of biomass at its origin is 0.8 and 0.2, respectively. This fraction decreases with increasing sludge age, i.e.  $X_d = f(\text{SRT}_{\text{aer}})$ . In Equations 5.5 and 5.6, it is assumed that the nitrogen content of cellular mass at generation averages 12.3% and that this content declines to 7% for the non-degradable fractions. The same values for phosphorus is assumed to average 2.6% and 1%. The assumed nutrient contents are typically higher than what often is found in pulp and paper activated sludge systems. However, these contents can be obtained from plant to plant by rather simple analyses. In contrast to using a traditional BOD:N:P ratio as the basis for nutrient dosage, Equations 5.5 and 5.6 provide great advantages as they take sludge age and biomass production into account.

Errors that have occurred when making mass balance calculations have usually been due to analytical errors in determining the total phosphorus and nitrogen in the sludge. Another problem can arise if nitrogen fixation is part of the biological processes. Phosphorus is only known to be discharged from the process with the effluent and with the excess sludge. Nitrogen discharge is slightly more complex. In the aerobic treatment of forest-industry wastewaters, it has been shown that if the dissolved nitrogen in the influent does not exceed 1.5-2 mg/l, microorganisms will obtain nitrogen from the air. If this is the case, the complexity of mass balancing will increase (Hynninen *et al.*, 1995).

## Organic load based control

Wastewater amount and BOD loading are fluctuating in a wide range so that optimisation of nutrient dosage is difficult. Changes of organic load may be caused by cyclic changes in the mill processes and partial close-downs. If effluent concentrations cannot be kept low enough by flow proportional control, a BOD load dependent dosage may be an alternative. Together with the flow rate, a signal representing the BOD load is generated. This signal can be used for controlling the dosage pumps. Control systems of this kind have been tested in Foster *et al.* (1999). Besides this feedforward control approach, the effluent residuals were measured and monthly mass-balances were made. The strategy resulted in significant reductions in nutrient usage, lower effluent nitrate levels and improved biomass quality.

The whole principle of feedforward control is simple; measure the disturbance (BOD load) and adjust the manipulated variable (dosage pumps) to compensate for the effects of the disturbance (Olsson and Newell, 1999). The BOD<sub>5</sub> test is a five-day analysis, which means that one does not know the true load to the plant until five days after the influent has entered the treatment system. For activated sludge systems with retention times of 24 hours, it is meaningless to adjust nutrient feed rates on last week's BOD<sub>5</sub> values (Foster *et al.*, 1999). If BOD can be computed from the

measured variable COD, and if the observed carbonaceous yield constant and the nutrient content of the sludge are known, the required amount of nutrients can be calculated.

The method especially suits nitrogen control, because the amount of nitrogen in the wastewater influent is usually low compared to the amount needed by the activated sludge process. The variations in the input of phosphorus (affected by the influent wastewater and return sludge) can be so large that organic load alone does not determine the need of phosphorus (Jansson *et al.*, 1994).

### Respirometry based control

A respirometer measures the respiration rate of a sample. The result is given as an OUR (oxygen uptake rate) curve. The integral (area) of this curve is a measure of the organic strength of the sample. Unlike the BOD<sub>5</sub> test, the initial biomass concentration in a respirometry test is high, permitting the test to be carried out fast. Normally samples are taken directly from the aerated tank with concentrations ranging between 4000 and 7000 mg COD/l. Some well-designed and robust respirometers are now commercially available (Ning *et al.*, 2000).

Respirometry is a way to directly measure how much nutrients is used by the biomass as it treats the waste. The equipment to do this automatically include an on-line respirometer and nitrogen and phosphorus sensors. The respirometer carries out its normal respirometric cycles until the biomass reaches an endogenous state. The nutrient requirement can then be calculated as the difference in nutrient concentrations before and after the test.

Ning *et al.* (2000) found that nitrogen deficiency within a sludge was manifested by an increase in endogenous respiration when adding nitrogen to the sample. In order to achieve a nutrient-balanced sludge while avoiding an overdose of nitrogen, a nitrogen deficiency quantification protocol using respirometry was presented.

## 6 Model development

In this chapter, some fundamentals of activated sludge modelling are presented. The model development, the real intellectual challenge of the work, is presented concurrently. At the end of the chapter, the extensions that have been made to the original Activated Sludge Model No. 1 are summarised. Finally, the extended model is presented using the traditional matrix representation.

### 6.1 Modelling activated sludge systems

#### Why models?

Models are used in the areas of science and engineering for several reasons. Within the field of wastewater treatment, a number of general purposes for mathematical models can be defined. Those given below have been listed in Jeppsson (1996).

- Design – models assist the design of time consuming and expensive experiments. They do not fully replace practical work but might reduce its extent.
- Research and education – models help researchers to develop and test hypotheses and thereby gaining new knowledge about the process. They are platforms into which all present knowledge of a process or system can be condensed, and thereby, they summarize what has been done and what needs to be done in subsequent works. For the students, models provide a tool to actively explore new ideas and improve the learning process.
- Process control – models allow for the development of new control strategies by investigating the system response to a wide range of inputs. The usefulness of this becomes obvious when studying a presently operating plant like the Hyltebruk WWTP as responses can be studied without endangering the plant.
- Forecasting – models are used to predict future plant performance when exposed to foreseen input changes.
- Performance analysis – models allow for analysis of total plant performance and simplifies troubleshooting and optimisation of details.

A common factor in the areas of usefulness given above is that the model is used to study a process of great complexity. Design, research and even controller tuning can be carried out successfully by analytical solutions or with the experience of the actor(s) if the system studied is applicable for these approaches. Biological WWT processes, however, are affected by a huge amount of reactions and environmental factors yielding long system time responses (a biological experiment may require months until it reaches steady state) and non-linear behaviour. Therefore, mathematical models are important, and in some contexts necessary, tools for studying the behaviour of biological systems.



## Different classifications of models

Models may be classified in many different ways. Here, four of them are mentioned.

Black-box, or statistical, models are based on empirical relationships between the system input and the system output. In contrast, a mechanistic (or white-box) model is based on the believed physical, chemical or biological mechanisms that affect the system. Of course, there are also models that are a mixture of the two extremes mentioned above, so-called grey-box models. In truth, all models of real processes, including the one presented in this work, are grey-box models. For example, the Monod form of the biological kinetic rate expressions used to model microbial growth is empirical in nature and could be replaced by a number of other expressions with equally good results. Still, the over all equation structures are based on fundamental mass balances (Olsson and Newell, 1999).

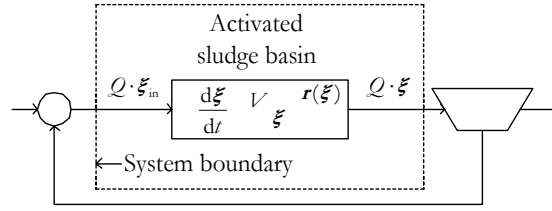
A model can be dynamic or static. Static models are often referred to as steady-state models and they model the equilibrium of the system. Conversely, dynamic models account for the time varying responses of the system. Dynamic modelling of a process, occurring in a biological/chemical reactor or system, forces accumulation mechanisms to be taken into account.

Models can also be divided into distributed or lumped-parameter models. Distributed models describe not only variations in time but also in space. Mathematically, this can only be described using partial differential equations resulting in complex simulation problems. A lumped-parameter approximation identifies regions in which composition, energy and momentum are approximately invariant with spatial dimension (Jeppsson, 1996). An example of such a region is a continuously stirred-tank reactor (CSTR) where the effluent component concentrations are assumed to equal the concentrations in the entire reactor. In practice, and in the case of activated sludge modelling, a distributed system such as a plug-flow reactor, is modelled by dividing it into a series of CSTRs.

A fourth fundamental classification that can be made is based on if the model equations are linear or nonlinear. The reactions occurring in wastewater treatment processes are rarely inherently linear. This induces that analytical solutions seldom exist and makes analysis more complex. The nonlinear characteristics of biological processes are exemplified in Section 6.3.2.

## General characteristics of activated sludge models

The activated sludge (AS) models described in this thesis are dynamic, lumped-parameter, grey-box models including nonlinear reaction terms. The model formulations are derived through component balances over a CSTR, see Figure 6.1, and give according to the requirement of continuity:



**Figure 6.1.** System boundary and notations for activated sludge basin models.

$$\underbrace{Q \cdot \xi_{in}}^{\text{in}} + \underbrace{r(\xi) \cdot V}_{\text{prod}} = \underbrace{Q \cdot \xi}_{\text{out}} + \underbrace{V \frac{d\xi}{dt}}_{\text{ack}} \quad (6.1)$$

Here,  $\xi$  and  $\xi_{in}$  are vectors of reactor and inlet concentrations (mg/l) of all modelled components,  $Q$  is the local volumetric inflow ( $\text{m}^3/\text{d}$ ),  $V$  is the reactor volume ( $\text{m}^3$ ) that is assumed to be constant and  $r(\xi)$  is a vector containing the reaction rates ( $\text{mg}/(\text{l} \cdot \text{d})$ ). In a static system, there is no accumulation term (the time derivative is set to zero) and the mathematical model becomes a set of algebraic equations.

$$0 = Q(\xi_{in} - \xi) + r(\xi) \cdot V \quad (6.2)$$

In a dynamic model of the same process, the accumulation terms are included and lead to a set of ordinary differential equations:

$$\frac{d\xi}{dt} = \frac{Q}{V}(\xi_{in} - \xi) + r(\xi) \quad (6.3)$$

Mathematically,  $\xi$  is called the state vector and contains the model states or components, the variables that uniquely determine the state of the process. If the reactor volume can be assumed to be constant, Equation 6.3 is a general description of activated sludge basin (and a variety of chemical/biological) models. Numerical algorithms normally reduce the dynamic model to a set of algebraic equations. These are more complex than the ones formed in the steady-state case and almost always require computer power to be solved. The simulation environment used in this thesis is described in Chapter 8 of the COST simulation benchmark manual (Copp, 2002) and is based on Simulink<sup>TM</sup>; a graphical user interface to MATLAB<sup>TM</sup>.

## 6.2 The Activated Sludge Model No. 1

In 1983, the International Water Association (IWA) formed a task group, which were to promote development, and facilitate the application of, practical models for design and operation of biological wastewater treatment systems. The goal was firstly to review existing models and secondly to reach a consensus concerning the simplest one having the capability of realistic predictions of the performance of single sludge systems carrying out carbon oxidation, nitrification and denitrification. The final result was presented in 1987 as the IAWQ Activated Sludge Model No.1 (ASM1) (Henze *et al.*, 2000). The model includes 13 components, which are affected by 8 dynamic processes (in this chapter, process will mean the conversion of a

component). Through material balances over a CSTR, 13 ordinary differential equations are derived.

Several versions and modifications of the original model have been developed since 1987. The Activated Sludge Model No. 2d (Henze *et al.*, 2000) was presented in 1999 and includes enhanced biological phosphorus removal (EBPR). Experiences from the ASM1 and ASM2d formed the bases for the Activated Sludge Model No.3 (ASM3) also presented in 1999 (Henze *et al.*, 2000). Still, the original ASM1 is probably the most widely used for describing WWT processes all over the world. The ASM1 has proved to be a reliable tool for modelling nitrification-denitrification processes and has initiated further research in modelling and wastewater characterization.

In Section 6.3 below, the ASM1 is described together with the model extensions developed in this work. Several nice descriptions of the ASM1 can be found in the literature. The outline of Section 6.3 follows Jeppsson (1996) and Henze *et al.* (2000).

### 6.3 Model extensions

The ASM1 was developed for carbon and nutrient removal of municipal wastewater. The C:N and C:P ratios are much lower for these waters compared to wastewaters from the pulp and paper industry and thus, they do not adequately describe rate-limiting effects of nitrogen and phosphorus on bacterial growth. Also, even though municipal treatment plants normally contain high levels of phosphorus and enhanced biological phosphorus removal becomes more and more common, the ASM1 does not take phosphorus removal into account. The traditional way to remove phosphorus is through chemical precipitation and EBPR significantly increases the complexity of a wastewater treatment model. This can be seen in ASM2d, which was developed to describe EBPR (Henze *et al.*, 2000). EBPR relies on anaerobic or anoxic zones within the WWTP. In contrast, the model presented in this work aims at model aerobic conditions. Therefore, the ASM1 was selected as basis for model development.

In many WWTPs treating industrial wastewater (including the Hyltebruk WWTP), phosphorus and nitrogen levels are so low that they affect microbial growth and the oxidation process. In order to accurately model these conditions, states and processes describing phosphorus and nitrogen in the system must be added. The ASM1 includes the conversions of nitrogen but lacks phosphorus state variables. Therefore, an ASM1 model, extended to include the latter, is presented below. This makes it possible to add nitrogen and phosphorus Monod functions limiting the aerobic growth of heterotrophs.

The physical, chemical and metabolic conversions of elements in a WWTP are complex and the phosphorus states and processes of the proposed model do not describe exactly what is happening within the plant. Hopefully, however, the result may qualitatively reflect the processes and more important, the result may form a basis for further development and model verification. For simplicity, and in accordance with literature, the fractionation of phosphorus presented below looks much like the fractionation of nitrogen applied in the ASM1.

The appearance of higher-order organisms, e.g. protozoa, in the Hyltebruk activated sludge plant is thought to reduce sludge production. This is associated with the regeneration of nutrients, which is of great importance in this work. Therefore, an extra COD state variable has been included to model the biomass of higher organisms.

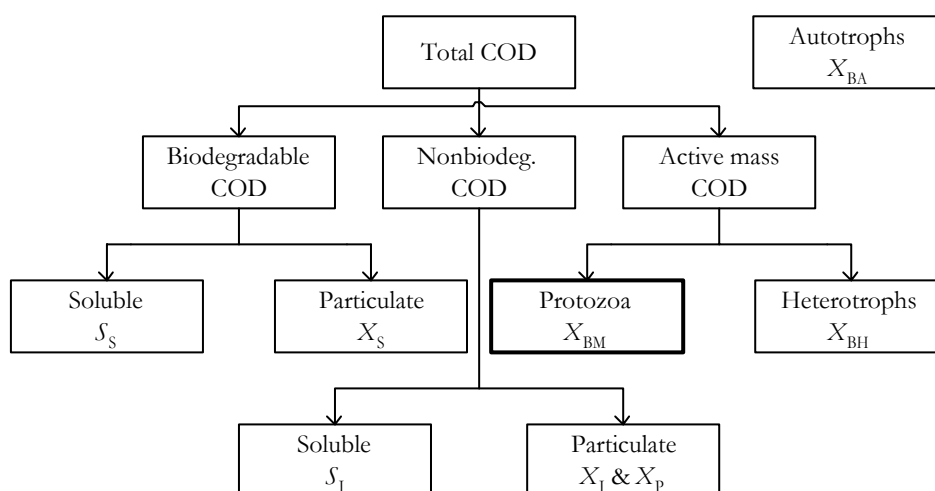
The ASM1 originally involves 8 processes incorporating 13 states. The fact that the Hyltebruk WWTP is run under strictly aerobic conditions (the oxygen concentration never falls below 1.5 mg/l) reduces the model formulation complexity as anoxic and anaerobic processes can be neglected. Also, the absence of nitrate nitrogen at the plant justifies neglecting the autotrophic biomass. The neutral pH-values make the modeling of alkalinity less important and therefore this state has been taken away.

To sum up, 3 dynamic processes and 3 state variables have been deducted from the original ASM1 model. In the model extension work, 3 processes and 10 state variables have been added. The extended model, presented below, therefore involves 8 processes incorporating 20 states. The importance of distinguishing between what had been done before and what is developed in this work is huge and therefore, the chapter ends with a section that summarizes the major extensions.

### 6.3.1 State variables

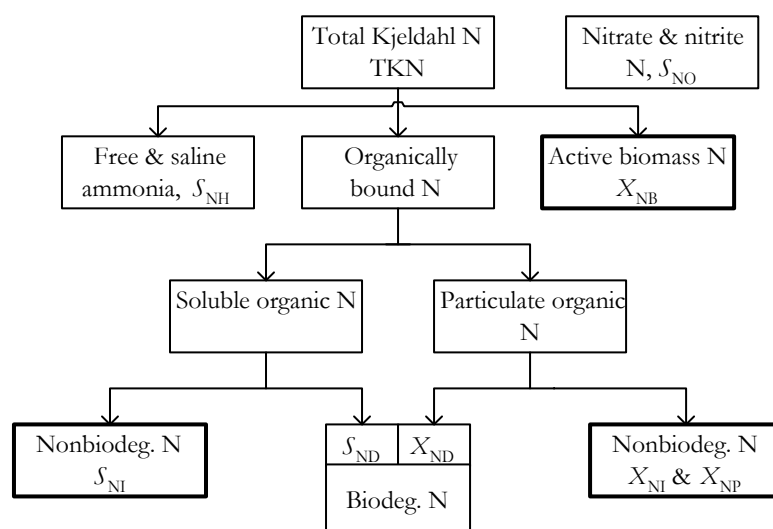
The presented model consists of 20 components or state variables that can be divided into COD components (mg COD/l), nitrogen components (mg N/l) and phosphorus components (mg P/l). Particulate components are denoted  $X$  and soluble components  $S$ .

The carbonaceous material in the ASM1 is divided into biodegradable COD, non-biodegradable COD (inert material) and biomass COD, see Figure 6.2. The readily biodegradable COD, which acts as substrate for aerobic heterotrophic growth, is denoted  $S_s$ . Particulate and soluble slowly biodegradable substrate are lumped together into one state,  $X_s$ . The non-biodegradable COD is divided into soluble ( $S_I$ ) and particulate ( $X_I$ ) inert material. Both are assumed to be unaffected by biological reactions in the system. The active biomass is divided into heterotrophic ( $X_{BH}$ ) and autotrophic ( $X_{BA}$ ) biomass. Growth of autotrophic biomass generates nitrate nitrogen not seen at the Hyltebruk WWTP so this component is not considered here. Finally, an extra state variable ( $X_p$ ) to model the inert particulate products arising from biomass decay is included. In the extended model, the influence of protozoa is accounted for by adding an extra active biomass state,  $X_{BM}$ . With the exception of this change, the carbonaceous material in the extended model is divided exactly as in the original ASM1 model.



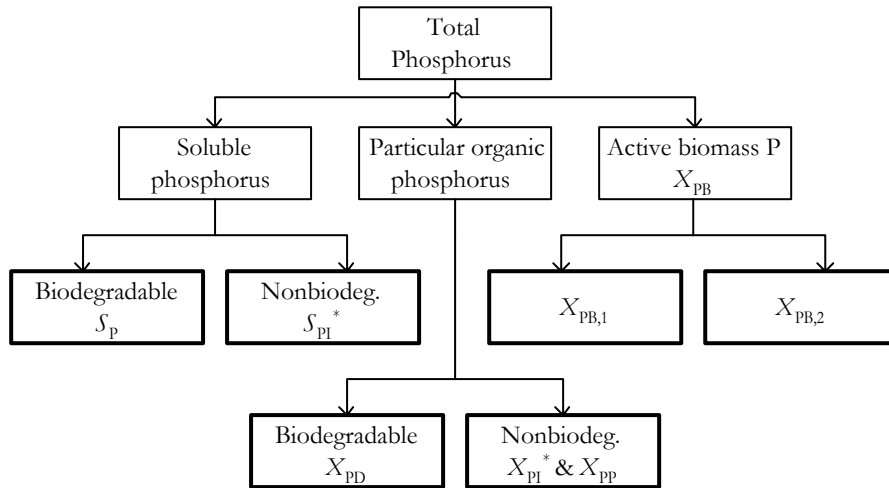
**Figure 6.2.** Wastewater characterization for carbonaceous components (from Jeppsson (1996)). Bold squares - added state variables. Dashed squares – deducted state variables.

Based on the total Kjeldahl nitrogen (TKN), the nitrogen is divided into free and saline ammonia ( $S_{NH}$ ), organically bound nitrogen and active biomass nitrogen, that is, the nitrogen fraction of the biomass, see Figure 6.3. Similar to the division of the organic material, the organically bound nitrogen is divided into soluble and particulate fractions, which in turn may be biodegradable or non-biodegradable. In the ASM1, only biodegradable particulate ( $X_{ND}$ ) and soluble ( $S_{ND}$ ) organic nitrogen are explicitly included in the model. Nitrate and nitrite nitrogen are lumped into one state,  $S_{NO}$ . The active biomass nitrogen ( $X_{NB}$ ), nitrogen in products arising from biomass decay ( $X_{NP}$ ), and nitrogen associated with inert particulate and soluble COD ( $X_{NI}$  &  $S_{NI}$ ) are surely present in a WWT system but are not explicitly included in the ASM1. There are two main reasons for this. Firstly, the ASM1 was developed for nitrification/denitrification processes. These are very common in municipal WWT plants and involve the conversion of nitrogen to gaseous ( $N_2$ ) form. This means that a component mass balance within the system would not be fulfilled even if taking all soluble and particulate N fractions into account. Secondly, the ASM1 claims that the states not included can indirectly be calculated from their associated COD states. This is only true if it can be assumed that the N content of the COD states are constant. As the extended model presented here primarily describes aerobic conditions (not producing nitrogen gas) and aims to take variations of nutrients in particulate material into account,  $X_{NB}$ ,  $X_{NP}$ ,  $X_{NI}$ , and  $S_{NI}$  are included as state variables.



**Figure 6.3.** Wastewater characterization for nitrogenous components (from Jeppsson (1996)).  
Bold squares - added state variables. Dashed squares - deducted state variables.

Much of the model development work in this thesis is concentrated on the inclusion of phosphorus components, not present in the ASM1. The total phosphorus is divided into soluble phosphorus, particulate organic phosphorus and active biomass phosphorus, see Figure 6.4. The soluble biodegradable form consists of inorganic orthophosphate and soluble organically bound phosphorus. Soluble organically bound phosphorus is hydrolysed to orthophosphate, the only fraction available for microbial growth. It is assumed that the hydrolysis occurs rapidly, in other words, that all non-inert soluble phosphorus becomes available for growth in a short time span. Thus the state variable  $S_p$  denotes soluble phosphorus available for assimilation. The inert part of the soluble phosphorus is denoted  $S_{pI}$ . Particulate organic phosphorus is divided into degradable and non-biodegradable particulate phosphorus. The degradable fraction is made up of particulate organically bound phosphorus ( $X_{pD}$ ). The nondegradable fraction consists of inert particulate phosphorus,  $X_{pI}$ , and particulate phosphorus arising from biomass decay ( $X_{pP}$ ). In order to describe additional phosphorus uptake, the living biomass is assumed to consist of two, from a model perspective, different forms of phosphorus: The first form,  $X_{pB,1}$ , is a constant part of the total biomass describing the minimum phosphorus content required for growth. The second form,  $X_{pB,2}$ , becomes a part of the biomass only when soluble phosphorus is present in excess. This 'luxury' uptake process is further discussed in Section 6.3.4. Phosphorus in municipal wastewaters is mainly present as phosphates in one form or another (Henze *et al.*, 1995). In industrial wastewater, however, inert phosphorus from production chemicals may be apparent. The inert phosphorus states  $S_{pI}$  and  $X_{pI}$  are not modelled explicitly (not included in the model) as they are thought to represent a very small part of the total phosphorus. However, the existence of these fractions should be considered.



**Figure 6.4.** Wastewater characterization for phosphorus components. All P states have been added to the original ASM1. States denoted with an asterisk are not explicitly included in the extended model.

### 6.3.2 Dynamical processes

In this section, the term dynamical process means the conversion of one component into another. In chemistry and chemical engineering sciences, these processes are normally referred to as reactions given by reaction rates. The transport (or flow) equations are also dynamic but as we assume completely mixed reactors, they affect the components in a straightforward way, and therefore, the main challenge when modelling AS systems is to describe the component conversions in an adequate way. Three main types of dynamic processes are considered here: microbial growth and decay, hydrolysis and ammonification.

#### Microbial growth and decay

Bacterial growth in activated sludge systems is the sum of two major processes: cell synthesis and cell decay. Cell decay is the natural death of bacteria combined with the cell maintenance requirements (the endogenous process), often expressed as a fraction of the biomass per day. Dynamic behaviour of each bacterial species can be described by three types of coefficients, namely the maximum specific growth rate  $\hat{\mu}$ , the decay rate  $b$ , and a set of half-saturation constants  $K_{i1}, K_{i2}, \dots, K_{in}$ . As will be discussed below, the number of half-saturation constants needed to model the growth depends on the number of growth-limiting components.

Both microbial growth and microbial death are, like most biological processes, autocatalytic reactions, which means that the catalyst is the product of the reaction. The growth rate ( $\mu$ ) of a biomass population  $X$  is defined as:

$$\mu X = r_{\text{growth}} \quad (6.4)$$

The specific biomass decay rate  $b$  is defined in the same simple manner:

$$bX = -r_{\text{decay}} \quad (6.5)$$

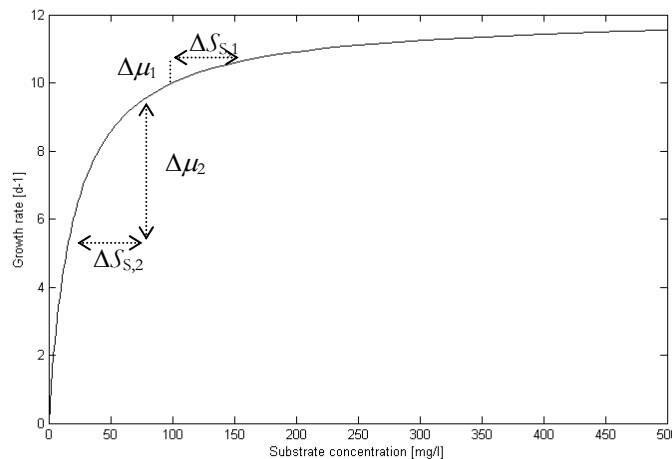
The observed net growth rate is thus:

$$r_{\text{net}} = (\mu - b)X \quad (6.6)$$

where both  $\mu$  and  $b$  have the unit  $\text{d}^{-1}$ . The growth rate can be influenced by a number of environmental factors. Normally, these influences are modelled using Monod kinetics. If the growth rate only depends on substrate concentration the observed growth rate  $\mu$  will become a function of the maximum growth rate  $\hat{\mu}$  and the substrate concentration  $S_s$ . If Monod kinetics are used, the equation becomes:

$$\mu = \hat{\mu} \left( \frac{S_s}{K_s + S_s} \right) \quad (6.7)$$

When the  $S_s$  concentration equals  $K_s$  the growth rate is half of the maximum growth rate, therefore the constant  $K_s$  is named the half-saturation constant. As  $S_s$  becomes high, the growth rate saturates towards  $\hat{\mu}$ . Naturally, the growth will cease as  $S_s$  becomes zero. The Monod functions give rise to the nonlinear characteristics of biological processes. Consider for example (see Figure (6.5)) a situation where bacteria grow on soluble substrate ( $S_s$ ) according to Equation (6.7). If the  $S_s$  concentration initially is high and changes to a lower, but still high, concentration the change in growth rate,  $\Delta\mu_1$ , will not be significant. If however, the substrate level is changed by the same amount from an initially more modest concentration, the change in growth rate,  $\Delta\mu_2$ , will be greater.



**Figure 6.5.** The nonlinear characteristics of the Monod function.

In the example given above, substrate was the only growth-limiting component. In reality many components and environmental factors can limit growth. These include temperature, alkalinity, toxics, pH and all nutrients. The limitation of several components on microbial growth can be described as follows:



$$\mu = \hat{\mu} \left( \frac{S_{n1}}{K_{S1} + S_{n1}} \right) \left( \frac{S_{n2}}{K_{S2} + S_{n2}} \right) \left( \frac{S_{n3}}{K_{S3} + S_{n3}} \right) \dots \left( \frac{S_{ni}}{K_{Si} + S_{ni}} \right) \quad (6.8)$$

where the indices 1 to  $i$  refer to the individual  $i$  nutrients  $n$ . In Equation 6.8, any single nutrient can be growth-rate limiting and so can all nutrients multiplicatively. In practice, however, only one of the components limits growth (Eckenfelder *et al.*, 1992).

In the ASM1, heterotrophic biomass is generated by growth on readily biodegradable substrate. The growth rate is limited by the concentrations of substrate and oxygen, which are modelled using Monod kinetics. In the extended model presented here, limitations of ammonia nitrogen and soluble phosphorus on heterotrophic growth are described by including Monod terms for these components into the process equation. This procedure is inspired by later versions of the ASM1 model, namely ASM2 and ASM2d (Henze *et al.*, 2000). The growth-rate equation for heterotrophic bacteria is modified to:

$$r_1 = \mu_H = \hat{\mu}_H \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left( \frac{S_P}{K_{P1} + S_P} \alpha + \frac{S_P}{K_{P2} + S_P} (1 - \alpha) \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) \quad (6.9)$$

The notation  $r_1$  refers to the order in which the process is presented in the model matrix (Section 6.3.6). There is an uncertainty about how low soluble phosphorus concentrations influence population dynamics and plant efficiency (Grau, 1991). Therefore it has been decided to model the limitations of soluble phosphorus as a sum of two Monod functions. The motivation for this approach is that it increases the flexibility of the shape of the net limiting function. The drawback is that the three additional parameters, the half saturation constants,  $K_{P1}$  and  $K_{P2}$ , and the weighting factor  $\alpha$  ( $0 \leq \alpha \leq 1$ ), have to be calibrated. In the model validation (Section 8.1) the three parameters have been chosen so that the net limiting function looks similar to a traditional Monod curve.

The growth-rate of protozoa is modelled similar to the growth-rate of bacteria. It is assumed that the growth-rate is limited by the concentration of heterotrophs (the ‘substrate’ for the protozoa), that the process requires oxygen and that these limitations can be described by Monod kinetics:

$$r_7 = \mu_M = \hat{\mu}_M \left( \frac{X_{BH}}{K_M + X_{BH}} \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) \quad (6.10)$$

It is an open question whether this approach is correct or not. Ratsak *et al.* (1995) suggest that a certain amount of bacteria is required to initialise protozoan growth. Equation 6.10 is one way to describe this. From a mechanistic point of view, the correctness of the rate-equation can be discussed. The value of  $\hat{\mu}_M$  and  $K_M$  can, however, be set so that the concentration of protozoa is realistic. Still, the dynamic behaviour of protozoa must be looked at critically.

Like in ASM1, decay of heterotrophs (Equation 6.11) is modelled using first-order kinetics. The heterotrophs decay at a certain rate and end up as inert particulate material or slowly

biodegradable substrate. The decay of protozoa (included in the extended model) is modelled in the same way (Equation 6.12):

$$r_2 = b_H X_{BH} \quad (6.11)$$

$$r_8 = b_M X_{BM} \quad (6.12)$$

The notations  $r_2$  and  $r_8$  refer to the order in which the processes are presented in the model matrix (Section 6.3.6).

## Hydrolysis

The hydrolysis process converts particulate/colloidal/complex organic molecules into small, readily degradable products available for bacterial growth. It may be a degradation of both particulate and dissolved solids. In the ASM1, hydrolysis processes are described as simple first-order processes with respect to heterotrophic biomass. Hydrolysis relies on the excretion of enzymes carried out by microorganisms. Therefore, the processes saturates as the amount of entrapped substrate becomes large in proportion to the biomass. As the heterotrophic bacteria require aerobic conditions, the hydrolysis rate is also limited by the oxygen concentration:

$$r_4 = k_h \left( \frac{X_s/X_{BH}}{K_X + (X_s/X_{BH})} \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) X_{BH} \quad (6.13)$$

The conversion of slowly biodegradable substrate to readily biodegradable substrate is associated with a conversion of particulate organic nitrogen and phosphorus to soluble forms. In the ASM1, it is assumed that the hydrolysis of entrapped nitrogen (Equation 6.14) is coupled with and occurs simultaneously with the hydrolysis of entrapped organics. In the extended model, it is assumed that particulate organic phosphorus is hydrolysed in the same way as COD and nitrogen. In contrast to the hydrolysis product of entrapped particulate organic nitrogen, the extended model states that no soluble organically bound phosphorus is formed. Instead, it is assumed that the hydrolysis directly produces biodegradable soluble (ortho-) phosphates (Equation 6.15). The hydrolysis rate of organic nitrogen and phosphorus depends on the ratio of nitrogen and phosphorus to the amount of slowly degradable substrate:

$$r_5 = r_4 \frac{X_{ND}}{X_S} \quad (6.14)$$

$$r_6 = r_4 \frac{X_{PD}}{X_S} \quad (6.15)$$

## Ammonification

Soluble organic nitrogen is converted to ammonia nitrogen through a second order reaction. The reaction is empirical in nature but has been found to be adequate for modelling the conversion when coupled to the process rate-equation for hydrolysis of entrapped organic nitrogen. The process is the same as in the ASM1:

$$r_6 = k_a S_{ND} X_{BH} \quad (6.16)$$

### 6.3.3 Model parameters

ASM1 consists of a number of parameters, which can be divided into kinetic and stoichiometric coefficients.

The stoichiometric coefficients are:

- the carbonaceous yield coefficients denoted  $Y$ . These were discussed in Chapter 3;
- a set of coefficients denoted  $i$  describing the nitrogen and phosphorus contents of the various COD variables and
- a set of coefficients denoted  $f$  describing the fraction of active biomass ending up as inert particulate material ( $X_p$ ).

The kinetic coefficients are:

- maximum growth rates ( $\hat{\mu}$ ) and decay rates ( $b$ );
- a set of half-saturation constants ( $K$ );
- hydrolysis rate ( $k_h$ ) and
- ammonification rate ( $k_a$ ).

The kinetic coefficients have the dimension  $d^{-1}$ , which means that a fraction of the biomass is affected per day. Besides the parameters included in the ASM1, a number of coefficients have been added to the extended model. The added kinetic coefficients describing the protozoan population are the maximum growth rate for protozoan  $\hat{\mu}_M$ , the decay rate of protozoa  $b_M$ , and the half saturation constant describing the heterotrophs' influence on the growth rate  $K_M$ . Half-saturation constants describing the growth limiting effects of nitrogen and phosphorus on heterotrophic growth have also been added. These have, when applicable, been chosen in accordance with ASM2d (Henze *et al.*, 2000) and with experiments described in Alexandersson *et al.* (2003b). Several stoichiometric coefficients have also been included in the extended model. These include the theoretical yield for autotrophs,  $Y_M$  and various coefficients for nutrient and phosphorus contents of the COD variables. A discussion of parameter values used in this work can be found in Chapter 8 and a summary of these is given in Appendix B.

### 6.3.4 Model formulation

Based on the above description, the full set of ordinary differential equations forming the extended Activated Sludge Model No.1 for a CSTR can be formulated. According to Equation 6.3, the 20 components make up a system of coupled differential equations:

$$\frac{d\xi}{dt} = \frac{Q}{V}(\xi_{in} - \xi) + r(\xi) \quad (6.17)$$

The conversion vector  $r(\xi)$  equals the transpose of  $\mathbf{S}$  (the stoichiometric matrix) times the vector of reaction kinetics for the 8 processes,  $\varrho(\xi)$ .

$$r(\xi) = \mathbf{S}^T \cdot \varrho(\xi) \quad (6.18)$$

The stoichiometric matrix  $\mathbf{S}$  and process rate vector  $\varrho(\xi)$  are found in Table 6.1. Not all 20 equations are presented below, but after reading this section, it should be clear how to use the full model matrix.

#### COD conversions

The dynamic behaviour of the protozoa biomass population is affected by growth and death. During energy transfer from bacteria to protozoa, energy is lost (from the system) due to inefficient biomass conversion. This is balanced with an associated oxygen demand. The biomass increases by cell growth and decreases by decay. The growth rate depends on the actual concentration of  $X_{BM}$  in a first-order manner. It also depends on food ( $X_{BH}$ ) and oxygen concentrations. The two processes can be seen as processes 7 and 8 in the model matrix. The total reaction rate is given by multiplication of the transpose of column 18 of the  $\mathbf{S}$  matrix with the process rate vector  $\varrho(\xi)$ :

$$r_{X_{BM}} = \hat{\mu}_M \left( \frac{X_{BH}}{K_M + X_{BH}} \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) X_{BM} - b_M X_{BM} \quad (6.19)$$

In the extended model, it is assumed that the nutrient contents of protozoa equal the contents of the heterotrophs. Therefore, the growth of protozoa is not limited by exogenous soluble nitrogen and phosphorus. In reality, it is possible that protozoa utilize exogenous nutrients instead of the nutrients present in the heterotrophs. However, if the fractions of nutrients are assumed to be the same for heterotrophs and protozoa, the net nutrient release/uptake will be the same. The protozoa present in the activated sludge reactor will decrease the sludge production as they involve an additional level in the ecological pyramid. This phenomenon might not be directly seen in Equation 6.19, instead it is incorporated in the reaction equation for heterotrophs.

According to the ASM1, the dynamic behaviour of the heterotrophic biomass population is affected by growth and death. In this extended model, predation by protozoa also reduces the heterotrophic biomass population. The three processes can be seen as processes 1,2 and 7 in the

model matrix. Ammonia and soluble phosphorus are used as the nitrogen and phosphorus source for cell synthesis. The growth is limited by substrate, nitrogen, phosphorus and oxygen:

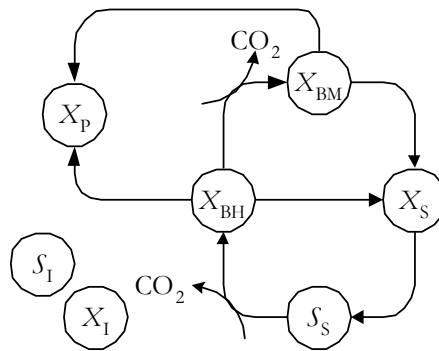
$$r_{XBH} = \hat{\mu}_H \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left( \frac{S_P}{K_{P1} + S_P} \alpha + \frac{S_P}{K_{P2} + S_P} (1 - \alpha) \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) - b_H X_H - \frac{\hat{\mu}_M}{Y_M} \left( \frac{X_{BH}}{K_M + X_{BH}} \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) X_{BM} \quad (6.20)$$

By comparing Equations 6.19 and 6.20 it is seen that one mass unit of heterotrophic bacteria generates only  $Y_M$  mass units protozoa. Since microbial energy efficiency is less than 100%,  $Y_M$  is less than one. Thus, the growth of protozoa leads to a decreased sludge production.

The concentration of soluble substrate is reduced by the growth of heterotrophic bacteria and produced by hydrolysis of slowly biodegradable substrate:

$$r_{SS} = -\frac{\hat{\mu}_H}{Y_H} \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left( \frac{S_P}{K_{P1} + S_P} \alpha + \frac{S_P}{K_{P2} + S_P} (1 - \alpha) \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) + k_h \left( \frac{X_S/X_{BH}}{K_X + (X_S/X_{BH})} \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) X_{BH} \quad (6.21)$$

Here as well, the negative COD production is seen by comparing Equations 6.20 and 6.21. In this process it is quantified by the stoichiometric coefficient  $Y_H$ . The decay of active biomass ( $X_{BH}$  and  $X_{BM}$ ) generates on the one hand slowly biodegradable substrate ( $X_S$ ) that is hydrolysed back to soluble substrate ( $S_S$ ) and on the other hand inert particulate products arising from biomass decay ( $X_P$ ). The conversion of COD states is illustrated in Figure 6.6. In Equation 6.21, note that the nitrate concentration is assumed to be zero.



**Figure 6.6.** The conversions of COD state variables.

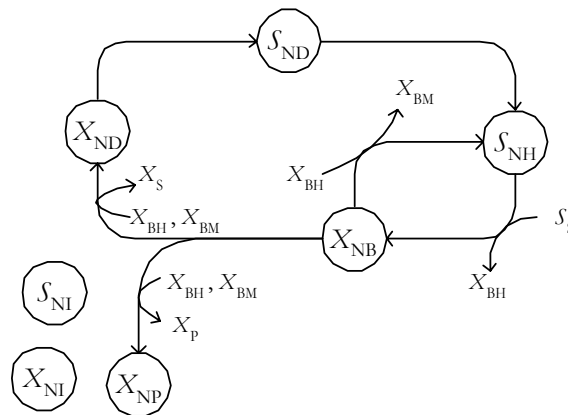
## Nitrogen and phosphorus conversions

To a large extent, the nitrogen and phosphorus components follow the same pathways as their corresponding COD states. When biomass is synthesized, soluble phosphorus and ammonia nitrogen are used up and converted to biomass phosphorus and nitrogen. The fraction of biomass ending up as inert particulate products from biomass decay is associated with corresponding N and P states ( $X_{PN}$  and  $X_{PP}$ ). The nitrogen and phosphorus state variables associated with the other fraction,  $X_S$ , are denoted  $X_{ND}$  and  $X_{PD}$ .  $X_{PD}$  can be hydrolysed directly into  $S_P$  while  $X_{ND}$  yields  $S_{ND}$ , which is converted to  $S_{NH}$  through the ammonification process. See Figure 6.7 and 6.8.

The reduced sludge production caused by the protozoan population leads to an interesting phenomenon with regard to the nutrients. Consider for example the conversions of ammonia nitrogen,  $S_{NH}$  in Equation 6.22. The first term represents production by ammonification of soluble organic nitrogen and the second term consumption through incorporation in heterotrophic biomass. The third term represents the predation of protozoa on the heterotrophs. This process converts only part of the heterotrophic biomass into new biomass. The nitrogen and phosphorus part of the heterotrophic biomass not used for synthesis of protozoan biomass will be regenerated to the solution and eventually be available for growth of bacteria:

$$r_{SNH} = k_a S_{ND} X_{BH} - i_{XB} \hat{\mu}_H \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left( \frac{S_P}{K_{P1} + S_P} \alpha + \frac{S_P}{K_{P2} + S_P} (1 - \alpha) \right) + \left( \frac{S_O}{K_{O,M} + S_O} \right) + i_{XB} \frac{1 - Y_M}{Y_M} \hat{\mu}_M \left( \frac{X_{BH}}{K_M + X_{BH}} \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) \quad (6.22)$$

As already mentioned, it is assumed that the nitrogen and phosphorus contents of protozoa become the same as the actual contents of the heterotrophs they consume. Protozoa might utilize exogenous soluble nutrients but in the figures covering the nitrogen and phosphorus conversions below, it is seen that the net flow of nutrients between heterotrophs and protozoa is modelled so that heterotrophic biomass N and P become protozoan biomass N and P.



**Figure 6.7.** The conversions of nitrogen state variables.

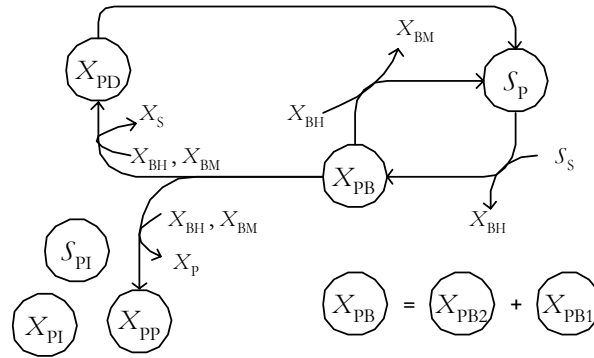


Figure 6.8. The conversions of phosphorus state variables

### Additional phosphorus uptake

In most cases, the nutrient contents of various COD components have been assumed to be constant. For example, the constant coefficients  $i_{XP}$  and  $i_{XPP}$  give the nitrogen and phosphorus contents of inert particulate material arising from biomass decay. This approach is in accordance with traditional activated sludge models, the ASM1 included. It is, however, well known that the nutrient contents of active biomass tend to vary depending on the amounts of nutrients available (e.g. Saunamäki, 1994). In conditions of high soluble nutrient concentrations, the biomass assimilates large amounts of nutrients into its cell structure while at conditions of low nutrient concentrations the organisms tend to survive on smaller amounts. In the literature, this phenomenon has been discussed mainly with regard to phosphorus while actually, the variable nutrient ratio of active biomass to some extent also include nitrogen (Verduyn, 1991).

The model presented here introduces two stoichiometric parameters describing the phosphorus-to-active-biomass ratio. The parameters influence two different phosphorus states, which together describe the total phosphorus content of the biomass. The behaviour of these states are illustrated in Figure 6.9 below where a fixed amount of active biomass is present in an AS basin. The growth rate is in balance with the decay rate and excess sludge removal so that the mass of active organisms is in steady state (i.e. constant).

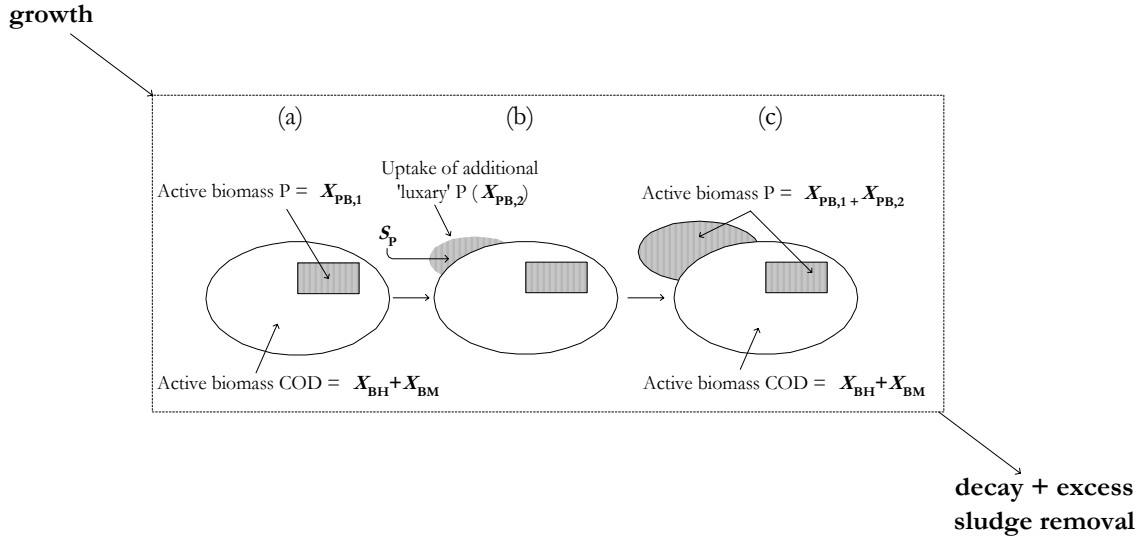
In (a), the AS process has been operated under phosphorus deficient conditions during some time. Still, the phosphorus amount is sufficient for microbial growth; the biomass takes up all exogenous soluble phosphorus so that the cell P content is minimal. This minimum amount of P required for growth is described with the state  $X_{PB,1}$ . It is given by the *constant* stoichiometric coefficient  $i_{XPB,1}$  so that:

$$\left. \begin{array}{l} \text{Mean content of phosphorus} \\ \text{in biomass at (a)} \end{array} \right\} = \frac{X_{PB,1}}{(X_{BH} + X_{BM})} = i_{XPB,1} \left[ \frac{\text{mg P}}{\text{mg COD}} \right] \quad (6.23)$$

In (b), the soluble influent phosphorus has been increased. When the environment provides a surplus of soluble phosphorus, the organisms take up more phosphorus than during starvation. This additional phosphorus is described with the state  $X_{PB,2}$ .

In (c), new steady-state conditions have appeared. The biomass P content is now made up of the minimum cell P content and the other additional phosphorus. The additional phosphorus is given by the *state dependent* stoichiometric coefficient,  $i_{XPB,2} = i_{XPB,2}(S_P)$  so that:

$$\left. \begin{array}{l} \text{Mean content of phosphorus} \\ \text{in biomass at (c)} \end{array} \right\} = \frac{X_{PB,1} + X_{PB,2}}{(X_{BH} + X_{BM})} \left[ \frac{\text{mg P}}{\text{mg COD}} \right] \quad (6.24)$$



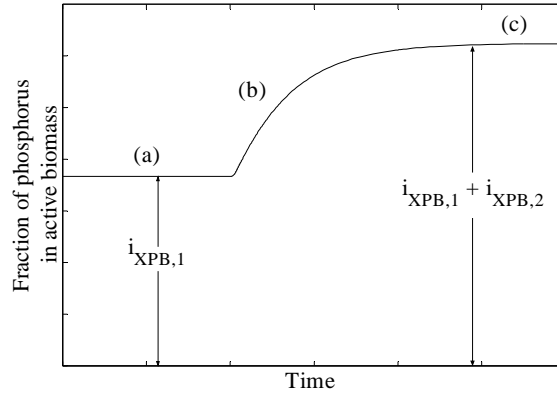
**Figure 6.9.** A schematic description of phosphorus uptake in the extended model. In (b), more soluble phosphorus becomes available and the 'luxury' phosphorus uptake process is switched on.

The coefficient  $i_{XPB,2}$  is a function of the soluble phosphorus concentration:

$$i_{XPB,2} = f(S_P) = \hat{i}_{XPB,2} \frac{S_P}{K_{ip} + S_P} \quad (6.25)$$

Two parameters determine the additional P uptake kinetics:  $\hat{i}_{XPB,2}$  is the maximum additional fraction of phosphorus (besides the required amount  $i_{XPB,1}$ ) a microorganism can assimilate. The additional P content of the organisms reaches this value as  $S_P$  remains high for a long time. The half-saturation constant  $K_{ip}$  determines how sensitive the accumulation process is with regard to the exogenous soluble phosphorus concentration. In Figure 6.10 below, the variations of  $i_{XPB,1}$  and  $i_{XPB,2}$  for the scenario (a)-(c) is illustrated.





**Figure 6.10.** An illustration of the *constant* stoichiometric coefficient  $i_{XPB,1}$  and the *state dependent* stoichiometric coefficient  $i_{XPB,2}$ . Cases (a), (b) and (c) are the same as in Figure 6.9. Total active biomass COD is in steady state.

Cells formed after an increase in available soluble phosphorus (like in scenario (b) above) will have a higher P content than the ones formed before the increase. The model cannot say anything about distributions of organisms with different P contents. Instead,  $i_{XPB,1}$  and  $i_{XPB,2}$  symbolize the mean P content of the total active biomass COD. A topic for future work could be to model the P uptake process so that it considers the distributions mentioned above.

According to Figure 6.8, soluble phosphorus ( $S_p$ ) is converted into active biomass phosphorus ( $X_{PB}$ ) as the heterotrophs synthesise new cells. As for ammonia ( $S_{NH}$ ), predation by protozoa is assumed to directly regenerate  $S_p$ . The decay processes form entrapped organic phosphorus ( $X_{NP}$ ), which can be hydrolyzed back to  $S_p$  and inert particulate phosphorus ( $X_{PP}$ ). The varying P ratio of active biomass makes the modelling of decay/predation processes that transform  $X_{PB}$  into other P components more complex. The obvious question that arises is ‘what is the P content of the presently decaying biomass?’

In this work, it is assumed that the release of P due to decay/predation processes is proportional to the mean cell P content. As protozoa decay (rate Equation 8), part of the active biomass P will be converted into inert particulate phosphorus ( $X_{PP}$ ). This part is given by the stoichiometric coefficient  $f_{PM}$ . The P content of this fraction is assumed to be constant  $i_{XPP}$ . The remaining P decay products are converted to entrapped organic phosphorus. As the total P decay products are assumed to be proportional to the mean cell P content, we conclude:

$$\left. \begin{array}{l} X_{PD} \text{ formed due to} \\ \text{protozoa biomass decay} \end{array} \right\} = \left( \frac{X_{PB,1} + X_{PB,2}}{X_{BH} + X_{BM}} - f_{PM} i_{XPP} \right) b_M X_{BM} \quad (6.26)$$

The following full dynamic rate equation of soluble phosphorus is an evident example of why the matrix format is preferred when describing complex models. The first term (the first two lines) describes the state dependent P uptake by formed biomass. The second term describes formation of soluble phosphorus by hydrolysis. Finally, the third term describes  $S_p$  production caused by the predation by protozoa.

$$\begin{aligned}
r_{SP} = & - \left( i_{XPB,1} + \hat{i}_{XPB,2} \frac{S_P}{K_{iP} + S_P} \right) \cdot \\
& \hat{\mu}_H \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left( \frac{S_P}{K_{P1} + S_P} \alpha + \frac{S_P}{K_{P2} + S_P} (1 - \alpha) \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) + \\
& k_h \left( \frac{X_S/X_{BH}}{K_X + (X_S/X_{BH})} \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) \left( \frac{X_{PD}}{X_S} \right) X_{BH} + \\
& \frac{X_{PB,1} + X_{PB,2}}{X_{BH} + X_{BM}} \frac{1 - Y_M}{Y_M} \hat{\mu}_M \left( \frac{X_{BH}}{K_{XBH} + X_{BH}} \right) X_{BM}
\end{aligned} \tag{6.27}$$

It is illustrative to compare the reaction 6.27 with the reactions of ammonia nitrogen (Equation 6.22). In contrast to nitrogen, phosphorus uptake by heterotrophs varies due to the state dependent coefficient  $i_{XPB,2}$ . The release of soluble phosphorus associated with protozoan predation is for phosphorus proportional to the *time-dependent* mean biomass P-fraction. For nitrogen, this fraction is constant.

### 6.3.5 Summary of model extensions and limitations

#### Model extensions

- An extra active biomass state variable,  $X_{BM}$ , has been added to model the influence of higher-order organisms. Autotrophic biomass has been deducted (Figure 6.2).
- Four nitrogen state variables, not explicitly modelled in the ASM1, have been added. Nitrate nitrogen has been deducted (Figure 6.3).
- A wastewater fractionation for phosphorus has been proposed. Five phosphorus state variables have been added (Figure 6.4).
- A traditional Monod-function is included to model ammonia nitrogen limitations on heterotrophic growth. A double Monod-function is included to model soluble phosphorus limitations on heterotrophic growth (Equation 6.9).
- The growth of protozoa has been modelled with Monod kinetics (Equation 6.10). The decay of protozoa is modelled similarly to decay of heterotrophs in the ASM1 (Equation 6.12).
- The extended model states that no soluble organically bound phosphorus is formed. Instead, it is assumed that hydrolysis instantly produces soluble phosphorus available for assimilation.
- The extended model presents two stoichiometric coefficients describing the phosphorus uptake rate. The first is a constant while the second is a state-dependent coefficient accounting for 'luxury uptake' (Figure 6.9, Equation 6.27).

## Model limitations/assumptions

- Oxygen concentration is not modelled.
- Alkalinity is not modelled.
- Nitrate and nitrite nitrogen is not modelled.
- Autotrophic organisms,  $X_{BA}$ , are not modelled.
- It is assumed that the hydrolysis of soluble organic phosphorus occurs rapidly, in other words, that all non-inert soluble phosphorus becomes available for growth in a short time.
- Protozoa are assumed to have the same nutrient content as heterotrophs.
- Nutrient release rate due to decay processes and predation are assumed to be proportional to the mean nutrient content of active biomass (Equation 6.26).
- The inert phosphorus states  $S_{PI}$  and  $X_{PI}$  are not modelled explicitly (not included in the model).

### 6.3.6 Extended model matrix

On the next two pages, the complete extended model matrix is presented. The stoichiometric matrix  $\mathbf{S}$  contains of the columns under the 20 components. The conversion vector  $r(\xi)$  is found right out on the right-hand side. The matrix representation allows rapid and easy recognition of the fate of each component. By moving down a column for a specific component, the full reaction rate may immediately be formulated and by moving across the matrix, the continuity of the model can easily be checked by calculating the sum of the stoichiometric coefficients.

$i$	Component $\xi \rightarrow$	1.	2.	3.	4.	5.	6.	7.	8.	9.	10.	11.	12.	13.	14.
$j$	Process $\downarrow$	$S_I$	$S_S$	$X_I$	$X_S$	$X_{BH}$	$X_P$	$S_O$	$S_{NH}$	$S_{ND}$	$X_{ND}$	$S_{NI}$	$X_{NB}$	$X_{NI}$	$X_{NP}$
1.	Aerobic growth of heterotrophs		$-\frac{1}{Y_H}$			1		$-\frac{1-Y_H}{Y_H}$	$-i_{XB}$				$i_{XB}$		
2.	'Decay' of heterotrophs				$1-f_P$	-1	$f_P$				$i_{XB} - f_P \cdot i_{XP}$		$-i_{XB}$		$f_P \cdot i_{XP}$
3.	Ammonification of soluble organic nitrogen								1	-1					
4.	'Hydrolysis' of entrapped organics		1		-1										
5.	'Hydrolysis' of entrapped organic nitrogen									1	-1				
6.	'Hydrolysis' of entrapped organic phosphorus														
7.	Aerobic growth of higher-order organisms					$-\frac{1}{Y_M}$		$-\frac{1-Y_M}{Y_M}$	$i_{XB} \frac{1-Y_M}{Y_M}$				$-i_{XB} \frac{1-Y_M}{Y_M}$		
8.	'Decay' of higher order organisms				$1-f_{PM}$		$f_{PM}$				$i_{XB} - f_{PM} \cdot i_{XP}$		$-i_{XB}$		$f_{PM} \cdot i_{XP}$
		Soluble inert organic matter [M(COD)L <sup>-3</sup> ]	Readily biodegradable substrate [M(COD)L <sup>-3</sup> ]	Particulate inert organic matter [M(COD)L <sup>-3</sup> ]	Slowly biodegradable substrate [M(COD)L <sup>-3</sup> ]	Active heterotrophic biomass [M(COD)L <sup>-3</sup> ]	Particulate products arising from biomass decay [M(COD)L <sup>-3</sup> ]	Oxygen (negative COD) [M(COD)L <sup>-3</sup> ]	NH <sub>4</sub> <sup>+</sup> + NH <sub>3</sub> nitrogen [M(N)L <sup>-3</sup> ]	Soluble biodegradable nitrogen [M(N)L <sup>-3</sup> ]	Particulate biodegradable nitrogen [M(N)L <sup>-3</sup> ]	Soluble inert nitrogen [M(N)L <sup>-3</sup> ]	Active mass nitrogen [M(N)L <sup>-3</sup> ]	Particulate inert nitrogen [M(N)L <sup>-3</sup> ]	Nitrogen in products arising from biomass decay [M(N)L <sup>-3</sup> ]

**Table 6.1.** The extended model matrix. All parameter values that have been used are given in Appendix B. The matrix continues on the next page.

	15.	16.	17.	18.	19.	20.	Process Rate $q(\xi)$ ↓
	$S_P$	$X_{PD}$	$X_{PP}$	$X_{BM}$	$X_{PB,1}$	$X_{PB,2}$	$\hat{\mu}_H \left( \frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left( \frac{S_P}{K_{P,1} + S_P} \alpha + \frac{S_P}{K_{P,2} + S_P} (1 - \alpha) \right) \cdot \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) X_{BH}$
	$-(i_{X_{PB,1}} + \frac{S_P}{K_P + S_P})$				$i_{X_{PB,1}}$	$\hat{i}_{X_{PB,2}} \frac{S_P}{K_P + S_P}$	$b_H X_{BH}$
		$\frac{X_{PB,1} + X_{PB,2}}{X_{BH} + X_{BM}} - f_P \cdot i_{X_{PP}}$	$f_P \cdot i_{X_{PP}}$		$-\frac{X_{PB,1}}{X_{BH} + X_{BM}}$	$-\frac{X_{PB,2}}{X_{BH} + X_{BM}}$	$k_a S_{ND} X_{BH}$
							$k_h \frac{X_S/X_{BH}}{K_X + (X_S/X_{BH})} \left( \frac{S_O}{K_{O,H} + S_O} \right) X_{BH}$
							$k_h \frac{X_S/X_{BH}}{K_X + (X_S/X_{BH})} \left( \frac{S_O}{K_{O,H} + S_O} \right) \left( \frac{X_{ND}}{X_S} \right) X_{BH}$
	1	-1					$k_h \frac{X_S/X_{BH}}{K_X + (X_S/X_{BH})} \left( \frac{S_O}{K_{O,H} + S_O} \right) \left( \frac{X_{PD}}{X_S} \right) X_{BH}$
	$\frac{X_{PB,1} + X_{PB,2}}{X_{BH} + X_{BM}} \frac{1 - Y_M}{Y_M}$			1	$-\frac{X_{PB,1}}{X_{BH} + X_{BM}} \frac{1 - Y_M}{Y_M}$	$-\frac{X_{PB,2}}{X_{BH} + X_{BM}} \frac{1 - Y_M}{Y_M}$	$\hat{\mu}_M \left( \frac{X_{BH}}{K_{X_{BH}} + X_{BH}} \right) \left( \frac{S_O}{K_{O,M} + S_O} \right) X_{BM}$
		$\frac{X_{PB,1} + X_{PB,2}}{X_{BH} + X_{BM}} - f_{PM} \cdot i_{X_{PP}}$	$f_{PM} \cdot i_{X_{PP}}$	-1	$-\frac{X_{PB,1}}{X_{BH} + X_{BM}}$	$-\frac{X_{PB,2}}{X_{BH} + X_{BM}}$	$b_M X_{BM}$
	Soluble biodegradable phosphorus [M(P)]L <sup>-3</sup>	Particulate biodegradable phosphorus [M(P)]L <sup>-3</sup>	Phosphorus in products arising from biomass decay [M(P)]L <sup>-3</sup>	Active protozoa biomass [M(COD)]L <sup>-3</sup>	Phosphorus in active biomass [M(P)]L <sup>-3</sup>		

Table 6.1. Continued from previous page.

## 7 Plant performance evaluation and influent wastewater characterisation

Application of the presented activated sludge model requires an influent fractionation of COD, nitrogen and phosphorus with regard to the model components and, therefore, the realization of a detailed measurement campaign is indispensable in a work like this. Also, wastewater analyses are used for model validation.

### 7.1 Measurement campaign

To determine the performance of the Hyltebruk WWTP, to derive mass balances and to achieve necessary information for model input fractionation, a measurement campaign was carried out during three days in May 2003. Eleven sample locations, illustrated in Figure 7.1 and described in Table 7.1 below, were selected:

Sample notation	Location/Description
in	Samples of mill wastewater entering the treatment plant collected at the Hyltebruk sample station. Nutrients have not been added to this water. The reject stream (water from sludge treatment) is included.
bs <sub>1</sub> , bs <sub>2</sub>	Samples collected in the two totally mixed suspended carrier biofilm reactors.
as <sub>1</sub> , as <sub>2</sub>	Samples collected from the two activated sludge basin effluents.
as <sub>1, sed</sub> , as <sub>2, sed</sub>	Samples of wastewater from the two secondary sedimentation unit effluents collected at the Hyltebruk sample station.
w <sub>1,1</sub> , w <sub>1,2</sub>	Samples collected directly from the two secondary settler underflows.
w <sub>2</sub>	Samples collected from the additional settler underflow. The samples taken here contain mixed sludge from the two lines.
eff	The WWTP plant effluent. It contains mixed wastewater from the two additional settler effluents. The samples were collected at the Hyltebruk sample station.

**Table 7.1.** Description of the sample locations chosen for the measurement campaign.

The stream notations bs, as, as<sub>sed</sub> and w<sub>1</sub> refer to mean values from the two lines. For example,  $S_{\text{COD,bs}}$  denotes the concentration of soluble COD calculated as a mean value based on analysis results from both lines.

During the three days, grab samples were taken once a day at each location. Samples from in, bs<sub>1</sub>, bs<sub>2</sub>, as<sub>1</sub> and as<sub>2</sub> were filtered through 1.6 µm GF/A papers within 20 minutes. After filtration the samples were put in cooling bags and transported to cold storage. Flow rates, dissolved oxygen concentrations, pH and temperature data were obtained from continuous sensors present at Hyltebruk.

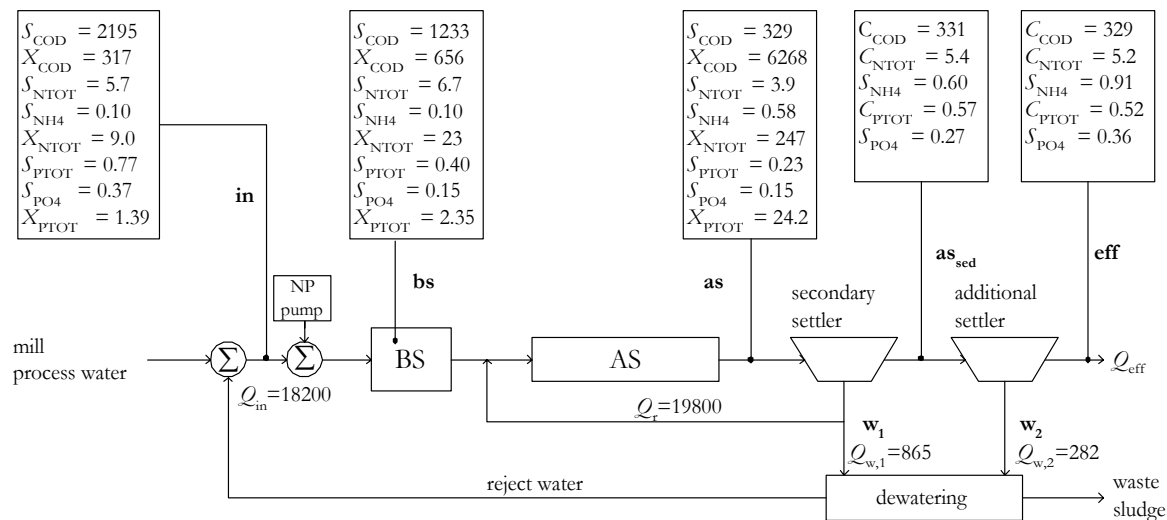
Analyses of a variety of parameters were carried out by Cenox AB. Filtered samples were analysed as total and filtered samples. Samples taken from the settler effluents (as<sub>1, sed</sub>, as<sub>2, sed</sub> and eff) and from the sludge streams (w<sub>1,1</sub>, w<sub>1,2</sub>, w<sub>2</sub>) were analysed as total samples only. The organic

content was analysed as COD, DOC, TSS and VSS. Nitrogen was analysed as total N (NTOT) and ammonium nitrogen ( $\text{NH}_4\text{-N}$ ). Phosphorus was analysed as total P (PTOT) and orthophosphate ( $\text{PO}_4\text{-P}$ ). All analyses were made according to standard methods except for total N, which was analysed using a forest industry method (SS-028101). Dissolved organic carbon (DOC) was measured on samples filtered through  $0.45 \mu\text{m}$  GF/A papers. All parameters were not analysed on each sample. In the complete result protocols (Appendix G), exact details of this can be found.

On the samples taken the second day,  $\text{BOD}_7$  analyses were made at all locations except for the three sludge streams. Also, a  $\text{BOD}_{28}$  analysis was made on the influent water (in). These analyses were also made according to standard methods. During the third day, three samples ( $as_{1-1}$ ,  $as_{1-2}$ ,  $as_{1-3}$ ) were taken from equally spaced locations along the activated sludge reactor of the first line. This was done in order to investigate the plug-flow characteristics of the activated sludge stage. COD, total N, ammonium, total P and DOC were analysed from these three samples.

During the test period, the total wasted excess sludge flow rates from the secondary settlers and additional settlers were kept constant at  $865$  and  $282 \text{ m}^3/\text{d}$ , respectively. The total plant inflow was  $18\,200 \text{ m}^3/\text{d}$ . The return sludge flow rate was  $19\,800 \text{ m}^3/\text{d}$  resulting in a recycle rate (Equation 2.1) of 110%. Dissolved oxygen concentrations were approximately  $3\text{-}3.5 \text{ mg O}_2/\text{l}$  in the biofilm basins and  $1.5\text{-}2.5$  in the activated sludge basins. The temperature was  $35^\circ\text{C}$  throughout the whole WWT system and pH ranged between 7 and 8.3.

The complete result protocols from the measurement campaign are found in Appendix F. In Figure 1, some mean values are illustrated. In the remainder of this chapter, the campaign results are analysed and treated in different ways. No significant or coherent differences between the two lines have been found and therefore, these are discussed together as mean values.



**Figure 7.1.** Stream notations (in bold) and mean values from the three-day measuring campaign.  $S$  denotes filtered samples,  $C$  total samples and  $X$  particulate samples.  $X=C-S$ . Note that the values at bs, as,  $as_{\text{sed}}$  and  $w_1$  are mean concentrations from the two lines. Values from the two sludge streams are not shown as they are believed to be invalid. The full protocols are found in Appendix G. Units: COD ( $\text{mg COD/l}$ ), N ( $\text{mg N/l}$ ), P ( $\text{mg P/l}$ ),  $Q$  ( $\text{m}^3/\text{d}$ ).

## 7.2 Steady-state calculations

To calculate COD-reductions, sludge age, sludge production and to track phosphorus throughout the plant, mass flows of total ( $C$ ), particulate ( $X$ ) and soluble ( $S$ ) COD and total P (PTOT) are calculated.

The total mass flows of component  $i$  in the inflow (in), in the flows from the biofilm stage (bs), in the effluent from the secondary sedimentation units ( $as_{sed}$ ), in the effluent (eff) and in the two sludge streams ( $w_1, w_2$ ) are computed as:

$$G_{i,in} = Q_{in} C_{i,in} \quad (7.1)$$

$$G_{i,bs} = Q_{in} C_{i,bs} \quad (7.2)$$

$$G_{i,assed} = (Q_{in} - Q_{w1}) C_{i,assed} \quad (7.3)$$

$$G_{i,eff} = (Q_{in} - Q_{w1} - Q_{w2}) C_{i,eff} \quad (7.4)$$

$$G_{i,w1} = Q_{w1} C_{i,w1} \quad (7.5)$$

$$G_{i,w2} = Q_{w2} C_{i,w2} \quad (7.6)$$

The fact that some analyses were made on both filtrated and total ( $C$ ) samples made it possible to distinguish between soluble ( $S$ ) and particulate ( $X$ ) components in these samples:

$$X_i = C_i - S_i \quad (7.7)$$

Mass balances for soluble and particulate total P and COD are calculated in the same manner as Equations 7.1-7.6.

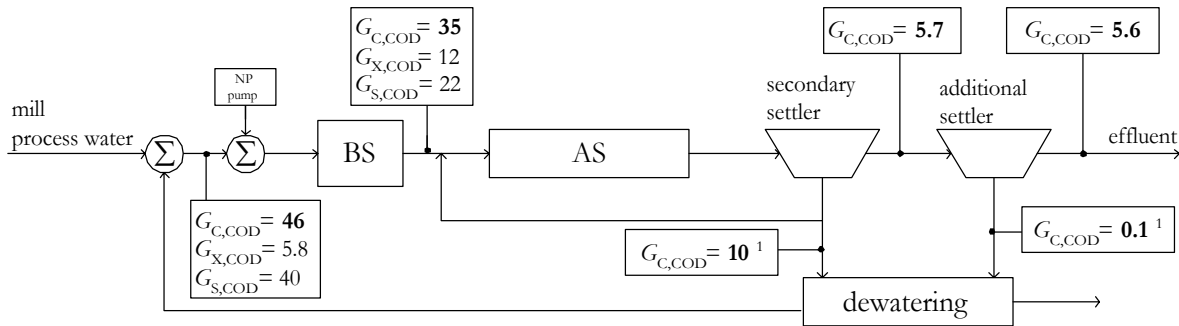
### 7.2.1 COD evaluation

To a large extent, the COD analyses show small variations with standard deviations within the margin of the measurement error. The TSS and VSS analyses (see Appendix E) made on water from the sedimentation processes show low particulate concentrations with VSS values ranging from 4 to 18 mg VSS/l. In Section 3.1, it was shown that 1 mg VSS approximately equals 1.4 mg  $X_{COD}$ . With regard to this, effluent total COD is associated with soluble COD.

However, the analysis results from the sludge flows are not representative as daily mean values. The VSS analyses of the secondary settler underflows indicate low values averaging 5140 mg VSS/l. The same parameter in the activated sludge basins averaged 4120 mg VSS/l. With a recycle rate of 110%, a settler mass balance shows that the VSS levels in the sludge flows must exceed the ones achieved from the analyses. This is discussed in detail in Appendix D. The low experimental values can be explained by means of the settler characteristics. Settlers are dynamic



units with time and space varying concentrations. At the time of sample collection, the sludge concentrations at the sample location might have been low. Also, the sludge concentration analyses made on the additional settler underflows were unreliable. To obtain reasonable values of the sludge concentrations, these were instead calculated from mass balances of the settlers, see Appendix D. The particulate COD concentrations of flow  $w_1$  and  $w_2$  were estimated to 12 000 and 1000 mg  $X_{\text{COD}}/l$ , respectively. If sludge concentrations are to be investigated in the future, it is proposed that samples are collected during 24 hours so that daily mean values can be analysed. With the campaign results and the approximated SS-concentrations in the sludge flows, the mass flows of COD throughout the system can be calculated using Equations 7.1-7.7. The results are illustrated in Figure 7.2 below:



**Figure 7.2.** Mass flows of total (C), particulate (X) and soluble (S) COD (ton COD/d) throughout the WWT plant. <sup>1</sup>Denotes that the value has been calculated.

COD reductions are calculated according to:

$$\left. \begin{array}{l} \text{Reduction of soluble COD} \\ \text{in biofilm stage} \end{array} \right\} = \frac{G_{\text{S,COD,in}} - G_{\text{S,COD,bs}}}{G_{\text{S,COD,in}}} = 0.45 \quad (7.8)$$

$$\left. \begin{array}{l} \text{Reduction of total COD} \\ \text{in biofilm stage} \end{array} \right\} = \frac{G_{\text{C,COD,in}} - G_{\text{C,COD,bs}}}{G_{\text{C,COD,in}}} = 0.25 \quad (7.9)$$

$$\left. \begin{array}{l} \text{Reduction of soluble COD} \\ \text{in activated sludge stage} \end{array} \right\} = \frac{G_{\text{S,COD,bs}} - G_{\text{S,COD,eff}}}{G_{\text{S,COD,in}}} = 0.41 \quad (7.10)$$

$$\left. \begin{array}{l} \text{Reduction of total COD} \\ \text{in activated sludge stage} \end{array} \right\} = \frac{G_{\text{C,COD,bs}} - G_{\text{C,COD,eff}}}{G_{\text{C,COD,in}}} = 0.64 \quad (7.11)$$

It is clear that a majority of the influent COD (87%) is associated with soluble COD. In the biofilm stage, mainly soluble, but also particulate, COD is converted to  $\text{CO}_2$  and particulate COD. The biofilm stage removes 45% of the influent soluble COD and of the 46 tons/d of total influent COD, 11.3 tons are oxidised to  $\text{CO}_2$ . The total biofilm stage COD reduction is consequently 25%.

The activated sludge stage removes approximately 41% of the influent soluble COD and 64% of the influent total COD. Of the 29 tons/d of COD removed in the AS stage one part is oxidised to  $\text{CO}_2$  and one part is removed with the excess sludge. It is estimated that each day approximately 10 tons of COD is removed with the excess sludge and that 19 tons of COD leaves the system as carbon dioxide. 5.6 tons COD/d leave the plant as inert soluble COD in the effluent. In total, the full plant reduces 86% of the influent soluble COD and 89% of the influent total COD.

## 7.2.2 Calculation of sludge productions and carbonaceous yield coefficients

The sludge productions ( $SP$ ) in the biofilm stage, in the activated sludge stage and for the whole plant are calculated using the mass flows of particulate COD. As discussed in Section 5.2, incoming sludge is deducted from the traditional  $SP$  expression:

$$SP_{bs} = (G_{X,COD,bs} - G_{X,COD,in}) = 6300 \text{ kg } X_{COD}/\text{d} \quad (7.12)$$

$$SP_{as} = G_{X,COD,w1} + G_{X,COD,w2} + G_{X,COD,eff} - G_{X,COD,bs} = -1800 \text{ kg } X_{COD}/\text{d} \quad (7.13)$$

$$SP_{tot} = G_{X,COD,w1} + G_{X,COD,w2} + G_{X,COD,eff} - G_{X,COD,in} = 4500 \text{ kg } X_{COD}/\text{d} \quad (7.14)$$

The observed yields for the biofilm stage, the activated sludge stage and for the whole plant are calculated as:

$$Y_{bs}^{obs} = \frac{SP_{bs}}{G_{S,COD,in} - G_{S,COD,bs}} = 0.35 \frac{\text{kg } X_{COD}}{\text{kg } S_{COD}} \quad (7.15)$$

$$Y_{as}^{obs} = \frac{SP_{as}}{G_{S,COD,bs} - G_{S,COD,eff}} = -0.11 \frac{\text{kg } X_{COD}}{\text{kg } S_{COD}} \quad (7.16)$$

$$Y_{tot}^{obs} = \frac{SP_{TOT}}{G_{S,COD,in} - G_{S,COD,eff}} = 0.13 \frac{\text{kg } X_{COD}}{\text{kg } S_{COD}} \quad (7.17)$$

The biofilm stage produces 6.2 tons  $X_{COD}/\text{d}$  and reduces 18 tons  $S_{COD}/\text{d}$ . The observed yield coefficient is therefore  $0.35 \text{ kg } X_{COD}/\text{kg } S_{COD}$ .

12.1 tons of particulate COD enters the activated sludge stage each day. During the same time, 10.3 tons of particulate COD are removed as excess sludge and 16.8 tons of soluble COD is reduced. The observed yield becomes  $-0.11 \text{ kg } X_{COD}/\text{kg } S_{COD}$ .

Each day, the whole plant produces 4.5 tons of COD as sludge and reduces 34 tons of soluble COD. The observed yield coefficient for the whole plant is therefore  $0.13 \text{ kg } X_{COD}/\text{kg } S_{COD}$ .

In Section 5.2, the aerobic sludge age were defined as the ratio between the total amount of aerated sludge and the amount of sludge leaving the plant. Here, it is calculated to 9.4 days.

		Biofilm stage	Activated sludge stage	Total plant
Reduction of soluble COD	%	45	41	86
Reduction of total COD	%	25	64	89
Sludge production	ton/d	6.3	-1.8	4.5
Observed yield coefficient	kg $X_{\text{COD}}$ / kg $S_{\text{COD}}$	0.35	-0.11 <sup>1</sup>	0.13
Sludge age	d	-	9.4 <sup>1</sup>	-

**Table 7.2.** Results from the COD removal evaluation and sludge production calculations.  
<sup>1</sup>Denotes that the value has been calculated from estimated sludge concentrations.

The results from the analyses made on samples taken along the first line of the activated sludge reactors are given in Appendix G. They indicate that the soluble COD is reduced soon after the pollutants have entered the basin. It is also seen that the MLSS decreases due to protozoan predation and biomass degradation along the reactors.

### 7.2.3 Phosphorus tracing

As discussed in Chapter 5, an efficient nutrient dosage control system should be developed with good knowledge of nutrient mass balances within the plant. Therefore, the measurement campaign results are used to track phosphorus throughout the system. With estimated sludge concentrations  $X_{\text{COD,w1}}$  and  $X_{\text{COD,w2}}$ , they are calculated as:

$$G_{\text{PTOT,w1}} = Q_{\text{w1}}(S_{\text{PTOT,w1}} + \bar{i}_{\text{X,COD,P}} X_{\text{COD,w1}}) \quad (7.18)$$

$$G_{\text{PTOT,w2}} = Q_{\text{w2}}(S_{\text{PTOT,w2}} + \bar{i}_{\text{X,COD,P}} X_{\text{COD,w2}}) \quad (7.19)$$

The mean content of nutrient  $n$  (N or P),  $\bar{i}_{\text{X,COD,n}}$ , in the different particulate COD components is calculated as:

$$\bar{i}_{\text{X,COD,n}} = \frac{X_{n\text{TOT}}}{X_{\text{COD}}} \quad (7.20)$$

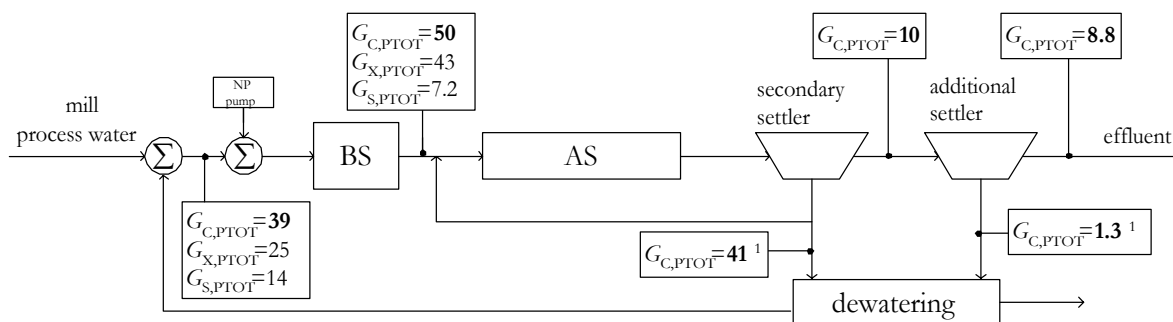
The N and P contents of the sludge were approximately 0.4% P and 4% N in the AS units. For details on this, see Appendix E where a variety of fractions have been calculated. As there are no reliable measurements of particulate material in the sludge flows, it is assumed that their relative N and P contents are the same as the contents in the activated sludge basins. The soluble nutrient concentrations in the underflow of a settler are often assumed to be equal to the concentrations in the overflow. Soluble phosphorus concentrations in the two sludge flows can therefore be calculated as:

$$S_{\text{PTOT,w1}} = C_{\text{PTOT,assed}} - \bar{i}_{\text{X,COD,P}} X_{\text{COD,assed}} \quad (7.21)$$

$$S_{\text{PTOT,w2}} = C_{\text{PTOT,eff}} - \bar{i}_{\text{X,COD,P}} X_{\text{COD,eff}} \quad (7.22)$$

As a majority of the particulate material is removed with the underflows of the settlers, the soluble P concentrations ( $S_{PTOT}$ ) are close to the total P concentrations ( $C_{PTOT}$ ).

The calculated mass flows of phosphorus and standard deviations are given in Appendix F. The mean values are illustrated in Figure 7.3 below:



**Figure 7.3.** Schematic outline of phosphorus mass flows (kg P/d) within the WWTP plant.  
<sup>1</sup>Denotes that the value has been calculated from mass balances over the settlers (Appendix D) and from the nutrient contents of the sludge in the activated sludge basins (Appendix E).

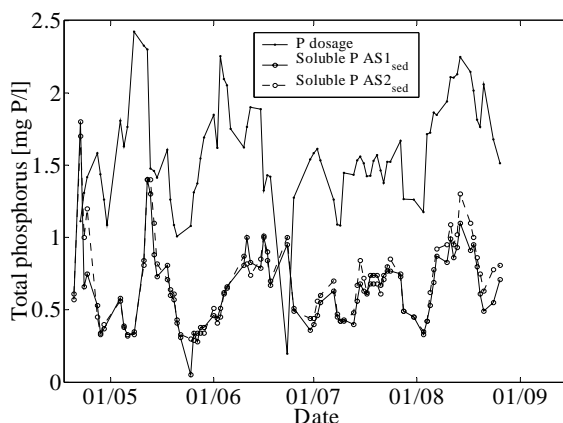
The calculations state that 39 kg P/d enters the WWTP with the influent. This P originates on the one hand from the up-stream mill processes and on the other hand from the reject water. In the biofilm stage, dosage of NP salt has increased the mass flow with 11 kg P/d to 50 kg P/d.

A majority of the phosphorus that enters the activated sludge stage is incorporated into particulate matter. 8.8 kg P/d, corresponding to a total phosphorus concentration of 0.5 mg P/l, leaves the WWTP with the effluent. Approximately half of this phosphorus is ortho-phosphate while the remainder is associated with the effluent TSS, soluble inert phosphorus and soluble organically bound phosphorus.

The amount of NP salt that should be dosed in order to fulfil the mass balance can be computed as:

$$G_{C,PTOT,dos} = G_{C,PTOT,eff} + G_{C,PTOT,w1} + G_{C,PTOT,w2} - G_{C,PTOT,in} \quad (7.23)$$

This calculation states that the dosage should be 13 kg P/d corresponding to a dosage concentration of 0.7 mg P/l. The reported dosages during the campaign time are higher, typically 20 kg P/d, and thus, what is going in to the system does not equal what is going out. This indicates that the phosphorus dosage at Hyltebruk is too high. As the total mass flow of P into the activated sludge stage is close to the total flow of P in the effluent and sludge streams, a possible phosphorus accumulation seems to occur in the BS. This phenomenon was also seen when dosage data from the Hyltebruk WWTP process computer system was studied. In Figure 7.4, the P-dosage is plotted together with total P-concentrations in the effluents of the secondary settlers.



**Figure 7.4.** P-dosage plotted together with total P-concentrations in the effluents of the secondary settlers.

The concentrations of soluble phosphorus in the effluent of the secondary settlers seem to be correlated with the dosed amount of phosphorus. The figure suggests that a time delay is present in the response of effluent phosphorus concentration due to dosage changes. This is most evident after the dosage increase during the end of May and could be the result of an accumulation phenomenon. According to the ratios calculated in Appendix E, the P content of particulate matter in the BS was not higher than 0.4%. A possible accumulation process therefore occurs in the biofilm of the carriers where no analyses were made.

## 7.2.4 BOD evaluation

From a BOD test, the biodegradable fraction of a wastewater is determined. For example, the standard BOD<sub>5</sub> test measures readily biodegradable, rapidly and some slowly hydrolysable organic matter whereas the ultimate BOD, BOD<sub>∞</sub>, measures additionally slowly hydrolysable organic matter and decay of biomass. The parameter was shortly summarized in Section 3.1. In the test, biomass is mixed with nutrients and a well-known amount of COD from the investigated pollutant. The mixture is then oxygenated and the bottle is closed. The oxygen consumption can then be measured as oxygen pressure changes in the bottle as a function of time. At the end of the test, oxygen and nutrients must be present in excess, otherwise these components might limit the oxidation process and the measurement result cannot be used.

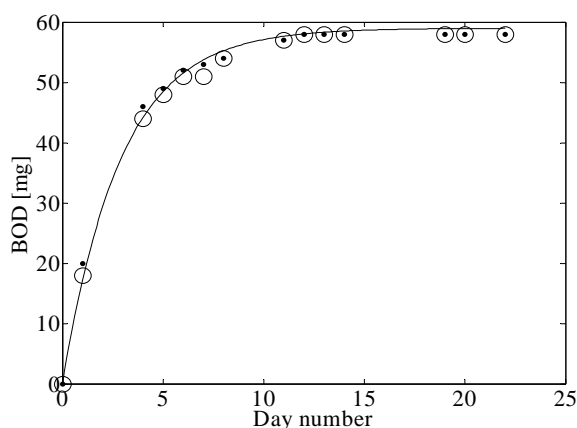
A BOD<sub>28</sub> test was carried out by Cenox AB on water from the Hyltebruk influent stream. The motive for making this test was to get information about the biodegradability of the wastewater. According to Swedish standards (SS-EN ISO 9408:1999/OECD 301F), biomass from a municipal treatment plant (Sjölunda WWTP, Malmö) was used. The laboratory set-up included 6 experiments running simultaneously. Besides two experiments carried out in accordance with the method described above, two experiments measured the biomass oxygen consumption, one controlled the activity of the biomass and one test was made to ensure that no toxic components inhibited the experiments.

The two pollutant samples were diluted to give a COD concentration of 100.3 mg COD/l. One litre of these samples were then mixed with water. The BOD was measured during 28 days and

the results can be seen in Figure 7.5. Normally, first-order kinetics are used to describe BOD curves (Weijers, 1999):

$$\text{BOD}(t) = \text{BOD}_\infty(1 - e^{-kt}) \quad (7.24)$$

The parameter  $k$  has been reported to range between 0.1 and 0.7  $\text{d}^{-1}$ . In this case, regression gives a  $k$  value 0.36  $\text{d}^{-1}$  and as can be seen, the kinetic model describes the data well.



**Figure 7.5.** Results from the BOD<sub>28</sub> test made by Cenox AB. The rings and dots refer to the duplicate tests made. Initially, the samples contained 100 mg COD. After day 12, the BOD value was stable.

In the above test, the BOD<sub>∞</sub> value was reached after approximately 12 days and amounted to 58 mg/l. The parallel test, which measured the oxygen consumption of the biomass, stated that 8 mg/l of this BOD was due to sludge activity. As the samples initially contained 100 mg COD, 50 % of the COD in the test substance was inert to microbial oxidation.

The COD tests made at different locations along the plant during the same period together with the experience of plant personnel indicate that the plant total COD reduction ranges between 85 and 90%, i.e. a significantly higher reduction than what the BOD<sub>∞</sub> states. It could be argued that particulate organic matter, removed in the settling processes and not biologically degraded, is part of this reduction and that this not is accounted for in the BOD<sub>∞</sub> value. However, one of the main results of the measurement campaign is that a majority, 87%, of the influent organic matter is made up of soluble COD. As environmental analyses, such as BOD tests, often are carried out due to government pressures, the main focus of the tests is to investigate the environmental impacts of the samples. Therefore it is standardized that municipal WWT biomass is used and that the experiment temperature should be 20°C. If this is the case, the BOD<sub>∞</sub> test can help answering two questions:

- If this pollutant is sent to the municipal WWTP, how much of it can be reduced?
- If this pollutant is discharged to the recipient untreated, how will it affect the recipient (probably lacking presence of acclimatized bacteria)?

The main reason for the differing reductions is that the BOD<sub>28</sub> test was carried out using biomass from a municipal treatment plant instead of biomass from the actual Hyltebruk plant. Bacterial

communities are formed by selection and, probably, the bacteria used in the test were not as efficient as those at Hyltebruk in reducing the existing pollutant.

There is a high probability that the acclimatized biomass at Hyltebruk, with protozoa present, degrades parts of the COD that were inert in the experiment. Also, the biological processes at Hyltebruk are running at 37°C and degradation efficiency increases with increasing temperature. The great difference between the BOD<sub>28</sub> test result and what actually has been observed at the plant forces the dismissing of the BOD<sub>∞</sub> value in this work. It is suggested that in future works, which aim to receive knowledge about a specific process, biomass from the actual treatment plant is used.

As for the BOD<sub>28</sub> test, the BOD<sub>7</sub> tests were made with biomass from a municipal WWTP. During 7 days, this biomass reduces 50% of the soluble COD in the wastewater. This is in accordance with the BOD<sub>28</sub> test, see Figure 7.5.

The BOD<sub>7</sub>/BOD<sub>∞</sub> ratio varies depending on the wastewater studied but normal values for raw municipal wastewater is 0.6-0.7 (Henze *et al.*, 1995). In this case it is 0.90, which means that 90% of the COD available for biomass growth has been reduced during the first seven days. This indicates that the biodegradable part of the COD is rapidly oxidised.

	in	bs	as	aS <sub>sed</sub>	eff
BOD <sub>7</sub> mg BOD/l	1300/1200	440	<8	12	9

**Table 7.3.** Results from the BOD<sub>7</sub> analysis in mg BOD/l. Note that the values at bs, as, aS<sub>sed</sub> and are mean values from the two lines. The analyses were made on total samples except for the ones made on influent wastewater, which were analysed as both total and filtered samples.

### 7.3 Influent wastewater characterisation

In this work, the Hyltebruk WWTP AS stage is modelled. The model input is therefore wastewater from the BS. The results from the fractionation are listed in Table 7.4.

The total COD is made up of soluble and particulate COD:

$$C_{\text{COD}} = S_{\text{COD}} + X_{\text{COD}} = S_{\text{I}} + S_{\text{S}} + X_{\text{I}} + X_{\text{S}} + X_{\text{BH}} + X_{\text{P}} + X_{\text{BM}} \quad (7.25)$$

Even though denoted with  $X$ , a fraction  $i_{\text{col}}$  of the slowly degradable substrate is soluble:

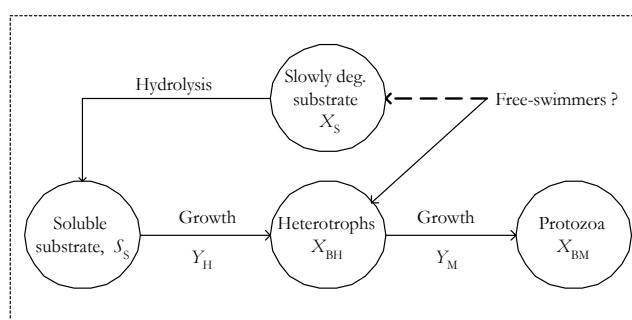
$$S_{\text{COD}} = S_{\text{S}} + S_{\text{I}} + i_{\text{col}}X_{\text{S}} \quad (7.26)$$

$$X_{\text{COD}} = X_{\text{I}} + (1-i_{\text{col}})X_{\text{S}} + X_{\text{BH}} + X_{\text{P}} + X_{\text{BM}} \quad (7.27)$$

The inert soluble COD,  $S_{\text{I}}$ , can be estimated with knowledge of the amount of COD not reduced in the plant. At Hyltebruk, the effluent COD rarely falls below 300 mg COD/l. The measurement campaign shows that the COD reduction takes place partly in the BS and partly in the very beginning of the activated sludge reactor. Thus, it is assumed that all remaining soluble

biodegradable COD is reduced in the AS stage and that COD in the effluent is associated with inert material. The effluent contains low levels of particulate material so this inert COD is assumed to be soluble, i.e.  $S_1 = 300$  mg COD/l. Since no information about  $i_{col}$  is available this constant is set to zero. Equation 7.26 then gives  $S_s = S_{COD} - S_1 = 933$  mg COD/l.

Regarding the particulate COD in the carrier stage, the model does not contain state variables to describe all of its components. Much of the particulate COD in the carrier stage is made up of free-swimming bacteria, a third (beside heterotrophic and protozoan) active biomass component. These are formed in the short hydraulic retention time environment of the BS and act as substrate for the protozoa in the subsequent AS stage. However, the protozoa also use the heterotrophs as substrate and in model terms, heterotrophic bacteria are the food for the protozoa. This was seen in Equation 6.10. In the extended model, no state variables describe the free-swimmers mechanistically. Consequently, the problem arises whether the active biomass in the carrier stage should be associated with  $X_s$  (slowly degradable substrate) or  $X_{BH}$  (active heterotrophic biomass), see Figure 7.6. A similar problem is discussed in the ASM1 model formulation (Henze *et al*, 2000), where the difficulty of determining the amount of heterotrophic bacteria in influents is considered. It is stated that this fraction can be assumed to be  $X_s$  if the yield coefficient  $Y_H$  is adjusted. Accordingly since the yield for protozoa,  $Y_M$ , is not known and instead estimated so that the protozoa concentration becomes realistic, the fractionation of free-swimming bacteria into  $X_s$  or  $X_{BH}$  can be made arbitrarily. It is decided to include them as slowly degradable substrate,  $X_s$ . Furthermore, it is assumed that no heterotrophs or protozoa are present in the AS influent:  $X_{BH}=0$ ,  $X_{BM}=0$ .



**Figure 7.6.** Two possible ways of modelling dispersed (free-swimming) bacteria. In this work it is decided to include them as slowly degradable substrate.

The particulate COD concentration of the activated sludge reactor averages 6 300 mg COD/l. Even if the system is operated with a long sludge age ( $\approx 9.5$  days), this amount of particulate material is rather high. A common way to achieve high MLSS model values is to assign particulate material in the influent to the inert  $X_1$  or  $X_p$  states. These components are then recycled in the system resulting in high particulate concentrations. However, the ratio of living biomass to particulate matter generated using this approach is much lower than what microscopy suggests. Therefore, a moderate level of inert particulate material is used in the influent:  $X_1=220$  mg COD/l and  $X_p=20$  mg COD/l. In order to increase the MLSS concentration, the decay and endogenous respiration rates are lowered. This adjustment is further discussed in Section 8.1. The resulting slowly biodegradable substrate concentration, now including free-swimmers, can be calculated by Equation 7.27:  $X_s = X_{COD} - X_1 - X_p = 436$  mg COD/l.



The fractionation of nitrogen and phosphorus is based on the particulate and soluble components:

$$C_{\text{NTOT}} = S_{\text{NTOT}} + X_{\text{NTOT}} \quad (7.28)$$

$$C_{\text{PTOT}} = S_{\text{PTOT}} + X_{\text{PTOT}} \quad (7.29)$$

All particulate COD-components are assumed to contain nitrogen and phosphorus:

$$X_{\text{NTOT}} = \sum (i_{\text{Xi}} X_i) \quad (7.30)$$

$$X_{\text{PTOT}} = \sum (i_{\text{XiP}} X_i) \quad (7.31)$$

As the amount of active biomass increases in the BS and AS stages, an increase in the  $X_{\text{NTOT}}/X_{\text{COD}}$  and  $X_{\text{PTOT}}/X_{\text{COD}}$  ratios throughout the WWTP is expected. In Appendix E, the fractions of nutrients to particulate COD at some sample locations have been calculated. The results are rather constant averaging 4% for N and 0.4% for P. With the available measurements, it is difficult to distinguish between the various fractions of nitrogen and phosphorus in different types of particulate material. It is assumed that the incoming inert particulate COD,  $X_{\text{I}}$ , has a N and P content of  $i_{\text{XI}}=3.5\%$  and  $i_{\text{XI,P}}=0.4\%$ , respectively. The N and P contents of formed inert particulate material,  $X_{\text{P}}$ , is assumed to be almost the same,  $i_{\text{XP}}=4\%$  and  $i_{\text{XP,P}}=0.4\%$ , respectively. Active biomass,  $X_{\text{BH}}$  and  $X_{\text{BM}}$ , is assumed to have a constant N content,  $i_{\text{XB}}=5\%$  N. The P content of active biomass is modelled so that it varies. The minimum constant P content of a living cell has been set to  $i_{\text{XBP,1}}=0.5\%$ . If soluble P is present in excess, it has been assumed that the P content can increase to 1%. The state dependent additional P content therefore varies between 0 and 0.5%, i.e.  $\hat{i}_{\text{XBP,2}}=0.5\%$ . The above assumed N and P fractions for the influent nitrogen states give:  $X_{\text{NI}}=8.8$  mg N/l,  $X_{\text{NP}}=0.7$  mg N/l and  $X_{\text{NB}}=0.1$  mg N/l. The influent phosphorus states become:  $X_{\text{PP}}=0.1$  mg P/l,  $X_{\text{PB,1}}=X_{\text{PB,2}}=0$ .  $X_{\text{PI}}$  is not explicitly included in the model but with a P content of inert material as assumed above,  $X_{\text{PI}}=i_{\text{XIP}} \cdot X_{\text{I}}=0.88$  mg P/l has been removed from the influent. The remainder N and P are associated with  $X_{\text{S}}$ . This gives  $X_{\text{ND}}=X_{\text{TKN}}-X_{\text{NI}}-X_{\text{NP}}-X_{\text{NB}}=13.4$  mg N/l, and  $X_{\text{PD}}=X_{\text{PTOT}}-X_{\text{PI}}-X_{\text{PP}}-X_{\text{BP1}}-X_{\text{BP2}}=1.4$  mg P/l.

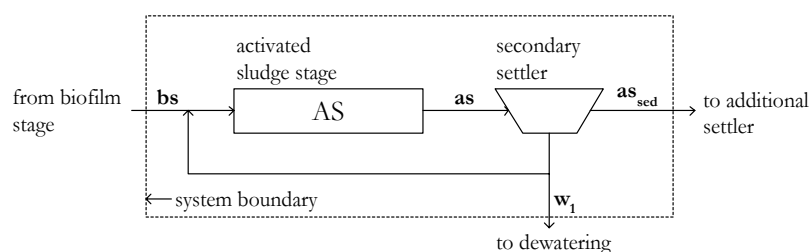
As for particulate components, soluble components also contain nitrogen and phosphorus. It is assumed that all soluble P and N are available for growth:  $S_{\text{NI}}=0$  and  $S_{\text{PI}}=0$ . Ammonia nitrogen and soluble biodegradable phosphorus is then directly available from the measurement campaign,  $S_{\text{NH}}=0.10$  mg N/l and  $S_{\text{P}}=0.40$  mg P/l. It is true that ortho-phosphate,  $S_{\text{PO}_4}$ , measured 0.15 mg P/l during the campaign. However, recall from Section 6.3.1 that all non-inert soluble phosphorus is assumed to be biodegradable. The remaining soluble nitrogen is associated with,  $S_{\text{ND}}=S_{\text{NTOT}}-S_{\text{NH}}=6.6$  mg N/l.

$j$				$j$			
1.	$S_I$	mg COD/l	300	11.	$S_{NI}$	mg N/l	0
2.	$S_S$	mg COD/l	933	12.	$X_{NB}$	mg N/l	0.1
3.	$X_I$	mg COD/l	220	13.	$X_{NI}$	mg N/l	8.8
4.	$X_S$	mg COD/l	436	14.	$X_{NP}$	mg N/l	0.7
5.	$X_{BH}$	mg COD/l	1	15.	$S_P$	mg P/l	0.4
6.	$X_P$	mg COD/l	20	16.	$X_{PD}$	mg P/l	1.38
7.	$S_O$	mg -COD/l	2	17.	$X_{PP}$	mg P/l	0.1
8.	$S_{NH}$	mg N/l	0.1	18.	$X_{BM}$	mg COD/l	1
9.	$S_{ND}$	mg N/l	6.6	19.	$X_{PB,1}$	mg P/l	0
10.	$X_{ND}$	mg N/l	13.4	20.	$X_{PB,2}$	mg P/l	0

**Table 7.4.** Results from characterization of the wastewater in BS.  $i_{XIP} \cdot X_I = 0.88$  mg P/l of the particulate phosphorus is assumed to be inert. The component numbers  $j$  are the same as in the model matrix (Section 6.3.6).

## 8 Model performance and validation

In Chapter 6, an extended activated sludge model was presented. In this work, the model is tested on the Hyltebruk activated sludge stage, including the secondary settler, see Figure 8.1.



**Figure 8.1.** System boundary for the simulated part of the Hyltebruk WWTP.

The extended AS model is implemented as C-files in MATLAB™/Simulink™. The settler is modelled with an existing 10-layer model (Takács *et al.*, 1991). The model assumes a constant cross-sectional settler area and is described in detail in for example Jeppsson (1996). The Hyltebruk AS reactors are designed to function in a plug-flow manner but since no trace element experiments have been carried out, the hydraulic characteristics of the reactors cannot be verified. However, simulations of the AS stage, both as one single CSTR and as a series of CSTRs, show similar results. It is decided to model one of the AS lines as a series of five equally sized CSTRs. The relatively high oxygen concentrations along the AS basin implies that oxygen do not influence the biological activity. In the model, a concentration of 2 mg –COD/l has been adapted. Further, it is assumed that the processes are operating at neutral pH and that kinetic parameter values for 30°C can be used. Some main characteristics of the simulated AS system are listed in Table 8.1 below.

Activated sludge characteristics			Settler characteristics		
$V$	m <sup>3</sup>	1500·5	Area	m <sup>2</sup>	1000
$Q_{in}$	m <sup>3</sup> /d	9100	Height	m	4
$Q_r$	m <sup>3</sup> /d	9900			
$Q_{w1}$	m <sup>3</sup> /d	433			

**Table 8.1.** Some main characteristics of the simulated AS system.

### 8.1 Selection of parameter values

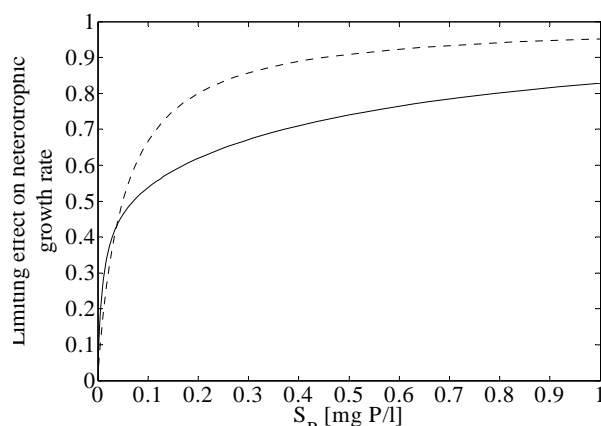
All parameters in the presented model must be given a value. This selection of values for the coefficients of a mathematical model is known as model calibration or parameter estimation. In the case of activated sludge models, model calibration requires extensive and well-controlled experiments carried out in pilot and/or bench-scale plants. Experiments of this kind lie beyond the scope of this work. Instead, reasonable parameter values for municipal WWTPs presented in the ASM1 and ASM2d model formulations (Henze *et al.*, 2000), to a large degree are used throughout the model performance evaluation. Some exceptions are discussed below. The choice of values for the new parameters, not included in ASM1 or ASM2d, are also presented and

motivated. The new or modified parameters are summarised in Table 8.2 below. All parameter values are found in Appendix B.

Some of the values given in the literature, such as growth and decay rates for heterotrophs, ammonification rate, hydrolysis rate and the half-saturation constant for hydrolysis of  $X_s$ , are temperature dependent following Arrhenius relations (Henze *et al.*, 2000). In this case, values at 30°C are used. It is true, that, the Hyltebruk AS stage is operating at approximately 35°C. The parameter values given in Henze *et al.* (2000) are, however, adapted for municipal wastewater and are not absolute values, especially not for pulp and paper wastewaters.

In order to maintain a high MLSS concentration in the AS stage, the endogenous respiration rate for heterotrophs is lowered significantly, compared to the value given in ASM1. Physically, this means that the organisms utilise less energy for maintenance and thus, less particulate matter is oxidised and the MLSS remains high. In the ASM1, as well as in the model presented here, this maintenance energy process is lumped together with the ‘traditional’ decay-rate, that is, the rate of cell-lysis. This is usually referred to as the death-regeneration hypothesis. In Figure 3.1, the decay and the endogenous respiration processes were separated but in model terms, the sum of the two processes are given by the lumped rate coefficient  $b_H$ . When simulating the Hyltebruk AS stage, this decay rate is lowered to 40% of the value given in the ASM1 formulation. This might be considered a drastic action, but if the MLSS concentration is to be kept high, and also contain a significant amount of active biomass, this seems to be the most reasonable way. Moreover, in ASM2d (Henze *et al.*, 2000), the decay rate has been lowered as well (see Table 8.2).

In Alexandersson *et al.* (2003b), it is concluded that the half-saturation coefficient for soluble phosphorus on aerobic microbial growth is not larger than 0.05 mg  $S_p$ /l. The ASM2d suggests that a single Monod-function with  $K_p=0.01$  is used. Here, a double Monod-function (see Equation 6.9) with  $K_{p1}=0.01$ ,  $K_{p2}=0.5$  and  $\alpha=0.5$  is applied. The function is similar to a traditional single Monod-function at low P-concentrations but during conditions of higher concentrations, the double Monod-function increases in a more linear manner than the traditional function.



**Figure 8.2.** Two different functions describing growth-limiting effects of soluble phosphorus on heterotrophic growth. Dashed line – a normal Monod-function with  $K_p = 0.05$ . Continuous line – the double Monod-function applied in the extended model with  $K_{p1}=0.01$ ,  $K_{p2}=0.5$  and  $\alpha=0.5$ .

The two terms included in the double Monod-function are weighted with the factor  $\alpha$ . They increase the flexibility of the shape of the net limiting function and it is proposed that the influence and usefulness of this new function is further investigated.

In accordance with Alexandersson *et al.* (2003b), the value for the half-saturation constant for ammonia nitrogen is chosen to  $K_{\text{NH}}=0.1$  mg N/l.

The kinetic coefficients for the protozoa ( $\hat{\mu}_M$ ,  $b_M$  and  $K_{\text{XBH}}$ ) are chosen so that they represent about 10-15% of the total MLSS. This is in accordance with microscopy carried out by personnel at Anox AB. The value of the protozoa yield coefficient is discussed in Section 8.3.1 below. The fraction of protozoan biomass yielding inert particulate products and the protozoan half-saturation constant for oxygen are chosen in accordance with the corresponding parameters for heterotrophs.

The nitrogen and phosphorus contents of various COD components are chosen so that they reflect the results from the measurement campaign. Their values were given in Section 7.4. In comparison with the values stated in ASM2d, especially the phosphorus contents have been lowered in the extended model. This is a typical difference between municipal and pulp and paper wastewaters. The half-saturation constant for additional phosphorus uptake,  $K_{\text{ip}}$  (see Equation 6.25), determines when the ‘luxury uptake’ process is switched on. The calibration of such parameters requires dynamic experiments. Such investigations have not been carried out and the selected value is therefore uncertain.

Extended model parameters	Symbol	Unit	Used value	ASM1/ASM2d (30°C)
<i>Stoichiometric coefficients</i>				
Protozoan yield	$Y_M$	g cell COD formed (g cell COD oxidised) <sup>-1</sup>	0.335	-/-
Fraction of protozoan biomass yielding particulate products	$f_{\text{PM}}$	dimensionless	0.08	-/-
Nitrogen:				
Mass N/mass COD in active biomass	$i_{\text{XB}}^*$	g N (g COD) <sup>-1</sup>	0.05	0.086/0.07
Mass N/mass COD in products from biomass decay	$i_{\text{XP}}^*$	g N (g COD) <sup>-1</sup>	0.035	0.06/0.02
Mass N/mass COD in inert particulate matter	$i_{\text{XI}}^*$	g N (g COD) <sup>-1</sup>	0.04	-/0.02
Phosphorus:				
Minimum mass P/mass COD in active biomass	$i_{\text{XBP},1}$	g P (g COD) <sup>-1</sup>	0.005	-/0.02 <sup>1</sup>
Mass P/mass COD in products from biomass decay	$i_{\text{XPP}}^*$	g P (g COD) <sup>-1</sup>	0.004	-/0.01

<sup>1</sup> This value is from ASM2d (Henze *et al.*, 2000) where the P content is assumed to be constant.

Mass P/mass COD in inert particulate matter	$i_{XIP}^*$	g P (g COD) <sup>-1</sup>	0.004	-/0.01
<i>Kinetic parameters</i>				
Protozoa:				
Protozoan max. specific growth rate	$\hat{\mu}_M$	day <sup>-1</sup>	1.2	-/-
Decay rate coefficient for protozoan organisms based on the death-regeneration hypothesis	$b_M$	day <sup>-1</sup>	0.1488	-/-
Half-saturation constant for heterotrophs	$K_{XBH}$	mg COD/l	7000	-/-
Half-saturation constant for oxygen	$K_{O,M}$	mg O <sub>2</sub> /l	0.2	-/-
Heterotrophs:				
Decay rate coefficient for heterotrophic organisms based on the death-regeneration hypothesis	$b_H^*$	day <sup>-1</sup>	0.744	1.86/0.8
Maximum additional mass P/mass COD in active biomass	$\hat{i}_{XBP,2}$	g P (g COD) <sup>1</sup>	0.005	-/-
Half-saturation constant for additional phosphorus uptake	$K_{iP}$	mg P/l	0.2	-/-
Half-saturation constant for soluble phosphorus	$K_{P1}$	mg P/l	0.01	-/0.01 <sup>2</sup>
Half-saturation constant for soluble phosphorus	$K_{P2}$	mg P/l	0.5	-/-
Weighting factor for $K_{P1}$ and $K_{P2}$	$\alpha$	dimensionless	0.5	-/-
Half-saturation constant for ammonia nitrogen	$K_{NH}^*$	mg N/l	0.1	-/0.05 <sup>3</sup>

**Table 8.2.** A summary of new and modified parameters. \*Denotes that the parameter is applied in ASM1 or ASM2d but that its value has been changed in the extended model. The full set of parameters is listed in Appendix B.

In Sections 8.2 and 8.3 below, some main results from simulations carried out with the parameter values listed in Appendix B are shown.

## 8.2 Steady-state simulations

In Table 8.3, steady-state results from a simulation of the Hyltebruk WWTP AS stage are shown. The values show good coherence with the measurement campaign results. It should be mentioned, however, that during this work, several different parameter sets giving approximately the same model behaviour and steady-state results have been found.

<sup>2</sup> ASM2d (Henze *et al.*, 2000) suggests a single Monod-function with  $K_P = 0.01$ . Alexandersson *et al.* (2003b) suggests that  $K_P$  (for a single Monod-function) is not higher than 0.05.

<sup>3</sup> ASM2d (Henze *et al.*, 2000) suggests that  $K_{NH} = 0.05$ . Alexandersson *et al.* (2003b) suggests that  $K_{NH}$  is not higher than 0.1.

The MLSS concentration has successfully been forced upwards. This is mainly due to the decrease in heterotrophic endogenous respiration rate but also due to the fact that  $X_{\text{COD}}$  in the AS influent has been increased by 22 mg COD/l (part of the influent particulate material (e.g. gravel) do not measure as COD, which, motivates this increase). The settler thickens the sludge in accordance with what was estimated in Appendix C.

The fractionation and parameter calibration effort mainly focuses on phosphorus and limited efforts have been made to fractionate the nitrogen. From the simulation results, it is seen that soluble nitrogen, present in excess, has been transformed into ammonia nitrogen. High levels of ammonia nitrogen were not detected in the measurement campaign. An explanation is that the soluble nitrogen found in the analysis results in reality is associated with inert soluble nitrogen, not available for ammonification.

The mean fraction of nitrogen in particulate matter is 4.3%. The same value for phosphorus is 0.46%. Soluble phosphorus in the AS effluent is not zero (0.14 mg P/l) but too low to initialise a significant additional P uptake. The mean phosphorus content of active biomass is therefore close to the minimal, 0.53%.

In the secondary settlers, particulate phosphorus is converted into soluble phosphorus. This might be due to anaerobic zones in the settler or significant cell decay. This phenomenon was mentioned in Section 5.3.2, where the problems with a single feedback control strategy was discussed. As biological or chemical processes in the settlers not are modelled, the phenomenon is not seen in the simulation results.

In the model, COD reduction is limited by ammonia nitrogen or soluble phosphorus. During the measurement campaign, nutrients for microbial growth, were available in excess and therefore, all biodegradable COD was oxidised. In terms of model behaviour, COD reduction will not decrease until one of the nutrients become limiting.

		bs (input)		as		w <sub>1</sub>				aS <sub>sed</sub>	
$S_{\text{COD}}$	mg COD/l	1233	(1233)	302	(329)	302	-	$C_{\text{COD}}$	mg COD/l	319	(331)
$X_{\text{COD}}$	mg COD/l	678	(656)	6408	(6268)	11768	(11930) <sup>1</sup>				
$S_{\text{NTOT}}$	mg N/l	6.7	(6.7)	5	(3.9)	5	-	$C_{\text{NTOT}}$	mg N/l	5.8	(5.4)
$X_{\text{NTOT}}$	mg N/l	23	(23)	274	(247)	504	-				
$S_{\text{NH}_4}$	mg N/l	0.1	(0.1)	4.8	(0.58)	4.8	-	$S_{\text{NH}_4}$	mg N/l	4.8	(0.60)
$S_{\text{PTOT}}$	mg P/l	0.4	(0.4)	0.14	(0.23)	0.14	-	$C_{\text{PTOT}}$	mg P/l	0.22	(0.57)
$X_{\text{PTOT}}$	mg P/l	2.35	(2.35)	29.2	(24.2)	53.3	-				
$S_{\text{PO}_4}$	mg P/l	-	(0.15)	-	(0.15)	-	-	$S_{\text{PO}_4}$	mg P/l	-	(0.36)

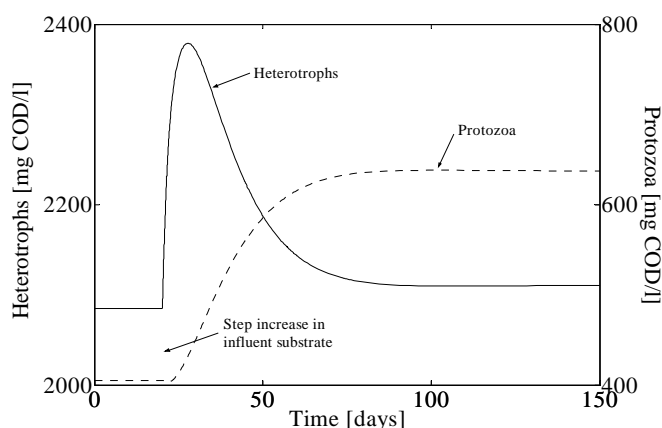
**Table 8.3.** Steady-state results when simulating the Hyltebruk AS stage with parameter values given in Appendix B. Mean values from the measurement campaign in parenthesis.  
<sup>1</sup>Denotes that the value has been calculated.

## 8.3 Dynamic simulations

Throughout the dynamic simulations described below, the parameter set listed in Appendix B has been used. As this work mainly focuses on limiting effects of phosphorus on biological activities, it is undesirable that nitrogen limits the COD reduction. Therefore, ammonia nitrogen in the influent has been kept high in the simulations.

### 8.3.1 Influence of protozoa

No literature values of protozoan kinetic parameters have been found. Instead, these parameters have been chosen so that the population represents about 10-15% of the MLSS concentration. This is achieved by reducing growth and decay rates compared with those of the heterotrophs. The theoretical yield coefficient for the protozoa was initially set to 70% of the heterotrophic yield. Even if these modifications cannot be verified, they are based on scientific reasoning. It is well known that higher-order organisms have slower kinetics (longer lifetimes) compared with more primitive organisms. Also, in comparison, they are thought to need more substrate for energy and thus have less for cell synthesis. In Figure 8.3, the slow dynamics of protozoa are illustrated. The net growth of heterotrophic bacteria has a time constant of about one or two days while the same value for protozoa is in the range of 20 days.



**Figure 8.3.** The slow dynamics of protozoa illustrated with a step increase in influent substrate at  $t = 20$  days.

The protozoa decrease sludge production as they involve an additional ecological level with associated energy losses. With  $Y_M = 0.7 \cdot Y_H$  the activated sludge net sludge production is -1500 kg/d. A 28% decrease of the protozoan yield coefficient (to  $0.5 \cdot Y_H$ ) decreases sludge production with 27% to -1900 kg/d. As discussed in Section 7.2.1, the sludge removal measurements are uncertain. However, they indicate a high sludge degradation (-1800 kg/d) and therefore,  $Y_M = 0.5 \cdot Y_H$  is used.  $\mu_M$  and  $b_M$  values used are approximately 10% of the corresponding rates for heterotrophs, see Table 8.2.

In this work, free-swimming bacteria formed in the biofilm stage are modelled as slowly degradable substrate,  $X_S$ . In the proposed activated sludge model processes, this substrate ( $X_S$ ) is



consumed by the floc-forming heterotrophs ( $X_{BH}$ ), which in turn are consumed by the protozoa ( $X_{BM}$ ). The fact that  $X_{BM}$  partly consume free-swimmers directly (without the intermediate  $X_{BH}$  level) is taken into account when calibrating the protozoa yield coefficient,  $Y_M$ . This was seen in Figure 7.6. As a consequence of modelling the free-swimmers as  $X_S$  as they enter the AS stage, the net sludge production of the AS stage is negative even when simulating the process without higher-order organisms. The reason for this is that the amount of biomass generated from influent soluble substrate falls below the amount of oxidised influent particulate matter. It is not obvious if this is likely to happen in reality. A physical explanation can be related to the long sludge age, a factor known to lower the observed carbonaceous yield. The presence of protozoa further increases sludge degradation and this is associated with a release of nutrients. Of the total net negative sludge production (-1900 kg/d) the protozoa account for -1000 kg/d. With a mean sludge P content of 0.4%, 4 kg P/d, which corresponds to 0.22 mg P/l in the AS influent, is released per day. In order to achieve complete reduction of biodegradable soluble COD, the model states that 0.25 mg P/l must be available in the AS influent. The same value without protozoa is approximately 0.47 (=0.25+0.22) mg P/l.

The appearance of protozoa results in a negative net sludge production and in theory, if the P contents of biomass in the BS and AS stages are assumed to be the same, the P content of biomass from the carrier stage suffices for COD reduction in the activated sludge stage. In reality, the plant probably does not perform satisfactory with a zero concentration of soluble nutrients in the AS influent. There are two reasons for this. Firstly, the carrier stage is totally mixed, and the AS influent equals the mixture in the BS. If soluble P or ammonia N concentrations are zero here, the concentration gradient required for biofilm diffusion is too small. Secondly, it is likely that not all nutrients assimilated into the free-swimmers are available for growth. A third problem is that too low nutrient levels may favour the growth of filamentous organisms (see Section 5.1) which, cause problems in AS.

In the model, the second reason mentioned above, causes influent soluble phosphorus concentrations to exceed zero for full oxidation of all biodegradable COD. It has been assumed that 0.88 mg/l of particulate phosphorus in the AS influent is inert. This leaves 1.38 mg P/l associated with the 436 mg/l slowly degradable substrate. This means that, in model terms, the free-swimmers and other particulate biodegradable substrates on an average contain only  $1.38/436=0.32\%$  bio-available P. The growth of biomass in the AS stage requires a minimum of  $i_{XBP}=0.5\%$  phosphorus. Therefore, even though the net sludge production is negative, soluble phosphorus must be found in the model input.

This is an evident example on how influent fractionation and parameter estimation determine the model outputs. If all influent particulate phosphorus was assumed to be available, no soluble phosphorus in the influent would be required for extensive COD degradation, and still, the net negative sludge production would produce soluble P in the effluent. The activated sludge reactor would act as a converter of particulate nutrients into soluble nutrients. An assessment of the bio-availability of particulate phosphorus in the Hyltebruk BS is surely one of the main tasks that must be investigated in order to develop a stable nutrient dosage control strategy.

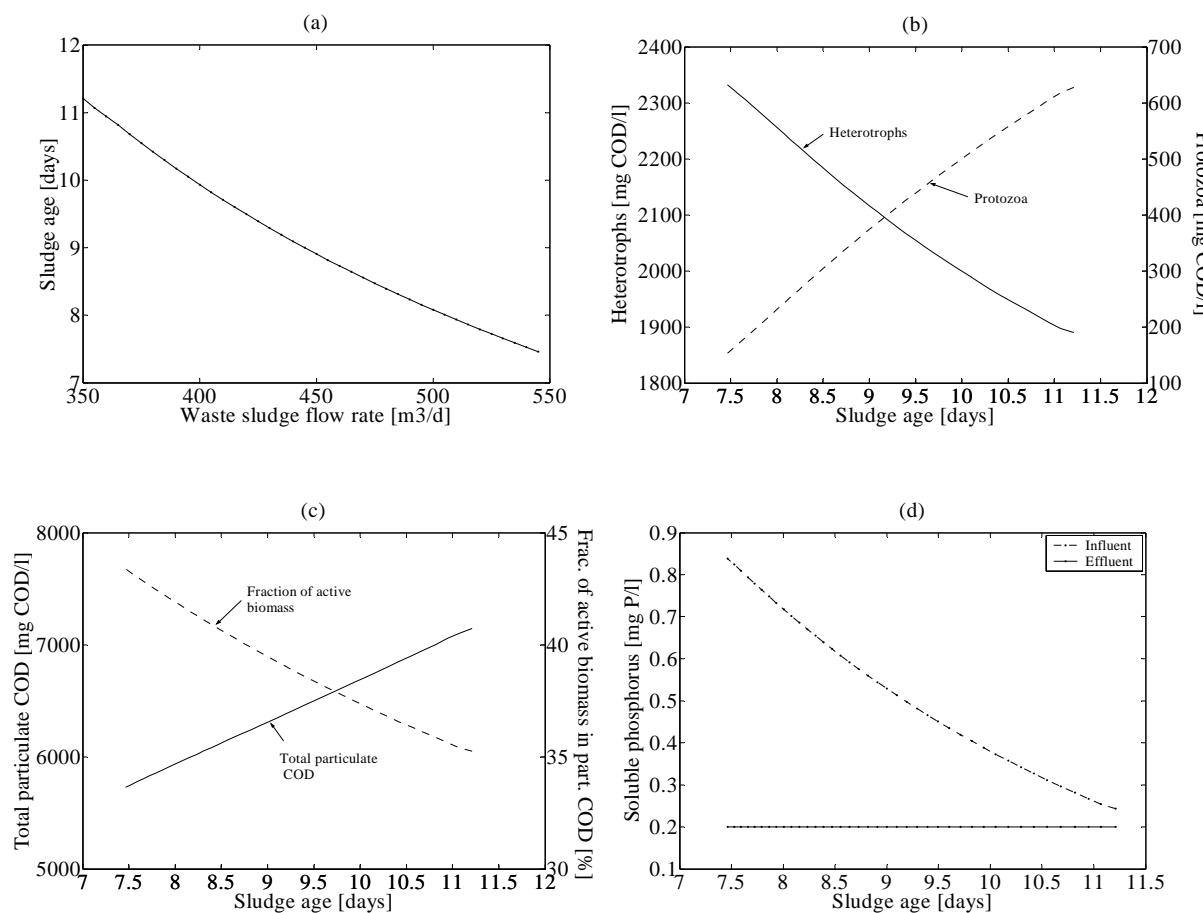
### 8.3.2 Influence of sludge age

The definition of sludge age and its influence on required influent nutrient concentrations was discussed in Section 5.2. By lowering the waste sludge flow rate ( $Q_w$ ), less sludge is taken out from the system and the MLSS concentration increases. If the oxygen concentration is sufficiently high, more particulate organic material will be oxidised. The result of this is lower sludge production (lower observed yield, see also Appendix H) and recycling of nutrients within the system. The phenomenon is reversed by the fact that the sludge concentration in the settler underflow ( $Q_w$ ) increases.

At this point, it should be clear that a lower observed yield means that appropriate pollutant reduction can be achieved with a higher BOD:P ratio in the influent. In the process modelled in this work, including protozoa, the influence of sludge age on nutritional requirements becomes even more apparent. The slow growth rate of protozoa (see Figure 8.3) makes them sensitive to sludge age. At longer sludge ages they have sufficient time for growth and compete with the heterotrophs (we are here assuming that protozoa settle in the same way as the heterotrophs), while at shorter sludge ages, they tend to die off or be washed out of the system. This phenomenon is similar to the ‘wash-out’ of autotrophs in nitrifying systems operating at short sludge ages. As already established, the presence of protozoa is an important factor when determining nutrient needs.

In Figure 8.4 below, some variables affected by sludge age are shown. The variable sludge age is simulated by changing the waste sludge flow rate  $Q_w$  (Figure 8.4a) and allowing the system to stabilise. A P-controller sets the influent soluble phosphorus concentration so that the effluent is maintained at 0.2 mg  $S_p$ /l (Figure 8.4d). The biodegradable soluble COD reduction is close to 100% at all sludge ages. At higher sludge ages, the protozoan population increases (Figure 8.4b), which lowers phosphorus requirements for appropriate treatment. The total particulate COD concentration increases when the sludge age increases as less sludge is removed from the system. However, the treatment efficiency (or oxidation rate) is not proportional to the increased MLSS concentration. This is due to an increased relative amount of inert non-active COD fractions (Figure 8.4c).

To summarise, an increased sludge age can lower the influent nutrient concentration needs. The drawback is an increased oxygen demand (as the sludge is oxidised instead of being wasted). Also, the sludge quality is influenced by the sludge age. This must be considered as it may affect the sludge treatment process and settling characteristics. In Section 7.2.2, the SRT at Hyltebruk was calculated to 9.4 days. In a control strategy, the SRT or waste sludge flow rate could possibly be a manipulated variable. However, changing the SRT influences the AS system over a period of days or weeks. A ‘rule of thumb’ is that 1-3 sludge ages of operation are required to establish and stabilise new process conditions. This means that for a WWT system with large and daily variations in influent characteristics, the SRT alone would not suffice for stable operation.



**Figure 8.4.** The influence of sludge age on nutritional needs.

In this context, it is relevant to look back at the steady-state calculations and measurement campaign. The total observed yield for the overall Hyltebruk WWTP is approximately 0.13 kg sludge COD produced/kg soluble COD reduced. In total, the plant removes 34 000 kg soluble COD per day and the mean nutrient content of the produced sludge is low, 4% N and 0.4% P (on a COD basis) throughout the entire plant. Consequently, the influent (mill process water + the additives) should bring approximately 18 kg P/d into the plant. However, the biofilm stage reduces 45% of the influent soluble COD and the yield for this stage is 0.35 kg sludge COD produced/kg soluble COD reduced. If this is to be maintained, 25.2 kg/d of soluble phosphorus must be found in the influent.

The conclusion is that the P requirements of the BS per day are 7.2 kg higher than for the plant in total. This corresponds to a soluble concentration of approximately 0.4 mg soluble P that theoretically will be found in the effluent if the plant is operated as during the measurement campaign. This phenomenon could be neutralised by lowering the sludge age. We want the total sludge production (Equation 7.14) to equal the sludge production in the BS (Equation 7.12):

$$Y_{\text{tot}}^{\text{obs}} \cdot (G_{\text{S,COD,in}} - G_{\text{S,COD,eff}}) = Y_{\text{bs}}^{\text{obs}} \cdot (G_{\text{S,COD,in}} - G_{\text{S,COD,bs}}) \quad (8.1)$$

i.e. the net sludge production in AS (Equation 7.13) is zero. According to the definition of sludge age in this work we conclude:

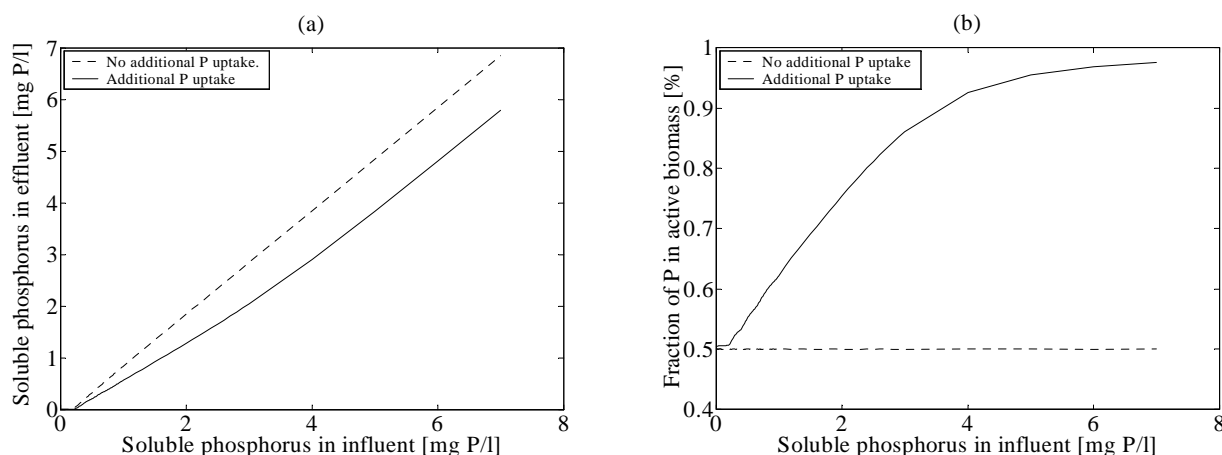
$$\text{SRT}_{\text{AS}} = \frac{V_{\text{AS}} \cdot X_{\text{COD,as}}}{Y_{\text{bs}}^{\text{obs}} \cdot (G_{\text{S,COD,in}} - G_{\text{S,COD,bs}}) + G_{\text{X,COD,in}}} \quad (8.2)$$

Using mean values from the measurement campaign, the ideal sludge age (Equation 8.2) of the AS would be 7.7 days corresponding to a total sludge outtake from the overall WWTP of 12 260 kg  $X_{\text{COD}}$ /d. If this was the case, soluble phosphorus in the BS and the effluent could both theoretically be zero. It should be noted that this steady-state calculation assumes that all the nutrients in the particulate material from the BS are available for the active biomass in the AS stage. This was not the case in the dynamic simulations, where  $0.88/2.75=32\%$  of the total influent phosphorus was assumed to be inert. According to the model, soluble phosphorus must be found in the influent even at a sludge age of 7.7 days.

Although based on a number of assumptions, the above analysis indicates that the Hyltebruk WWTP would benefit from a lower SRT in the activated sludge stage. A drawback with a decreased sludge age is that the sludge production increases. Furthermore, instead of decreasing the P-requirements for the BS from 25 to 18 kg P/d, this approach increases the P-requirements of the overall plant from 18 to 25 kg P/d. A second solution would be to decrease the P-requirements of the BS. It has been reported that this is possible but that it involves production of slime-forming bacteria in the BS. These then seem to propagate unaffected throughout the AS and cause increased TSS concentrations in the effluent.

### 8.3.3 Influence of additional phosphorus uptake

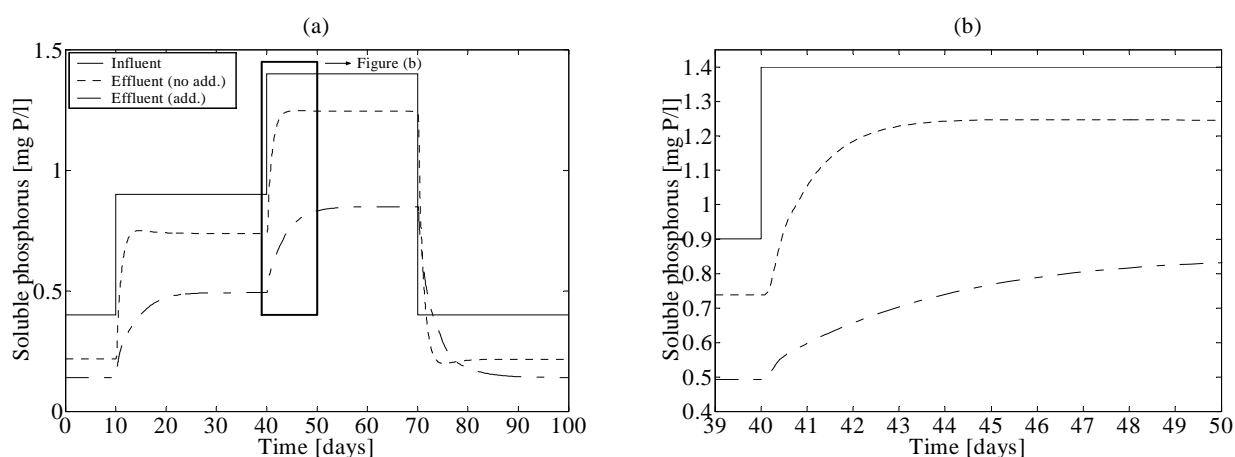
Below, variations in incoming soluble phosphorus concentrations have been simulated with and without the additional phosphorus uptake process. The latter scenario is obtained by setting  $i_{\text{XPB,2}}^{\wedge}$  (Equation 6.25) to zero. In Figure 8.5a, when all soluble biodegradable COD has been oxidised (at  $S_{\text{P,in}} \approx 0.22$  mg P/l), the concentration of effluent soluble phosphorus increases in a linear fashion as a function of the influent soluble phosphorus concentration. With the additional P uptake process switched on, the 'luxury uptake' also is initialised when soluble phosphorus is available in excess, i.e. when full reduction of biodegradable COD has been achieved.



**Figure 8.5.** Effluent soluble phosphorus concentrations and phosphorus assimilated by the active organisms as a function of influent soluble phosphorus.

However, the effluent P concentration increases in a different manner. Firstly, the formed biomass takes up more P then during conditions of P-limitation. Secondly, when the biomass contains a maximum of phosphorus, effluent soluble phosphorus begins to increase linearly in relation to the influent soluble phosphorus concentration. In Figure 8.5b, it is demonstrated that the fraction of P in active biomass without the additional P uptake process switched on is constant, or exactly  $i_{XPB,1}=0.5\%$ . Assimilation of additional phosphorus allow the P fraction to range between 0.5% and 1% ( $i_{XPB,1}+i_{XPB,2}$ ). The amount of active biomass P approaches its maximum at  $S_{p,in}>4$  mg/l.

The dynamics of the additional phosphorus uptake process is simulated in Figure 8.6 below. The time constant for soluble effluent phosphorus due to a step change in influent soluble phosphorus is without ‘luxury uptake’ one or two days, i.e. approximately the hydraulic retention time. The changes of biomass structure, generated by variable P uptake, increases the time constant to about eight or nine days, i.e. the sludge age. Further, the gain of the step response is lowered by the additional phosphorus uptake process.



**Fig 8.6:** The responses of effluent soluble phosphorus concentration as a result of step changes in influent soluble phosphorus concentration depend on the additional phosphorus uptake process.

Several parameters influence the shape of the step response and no great effort has been made to calibrate them for the Hyltebruk WWTP conditions. However, by comparing Figure 8.6 above with Figure 7.4 (where the reported P dosage and observed effluent soluble phosphorus were plotted), it is clear that the simulation results are reasonable. When the dosage is ranging between 0.4 and 1.4 mg P/l (which corresponds to 7.3 and 25.5 kg P/d), the soluble phosphorus in the AS effluent ranges between 0.2 and 0.85 mg P/l.

It could be argued that the possible time delays in the ‘true’ data, presented in Figure 7.4, are a result of accumulation in both the BS and AS stage. Also, the assumed accumulation processes seen in the measuring campaign seemed to appear in the BS. However, all phosphorus that possibly is accumulated in the BS propagate to the AS stage (i.e. becomes the model input) with the result that the time delays become larger than if the accumulation only occurred in the AS stage.

## 9 Conclusions

The conclusions presented in this chapter are based partly on the literature survey (Chapters 3 and 5), partly on the results of the measurement campaign that was carried out (Chapter 7) and partly on results from simulations of the developed activated sludge model (Chapters 6 and 8).

The nitrogen and phosphorus content of organic matter do not suffice for cell synthesis and as a result, these nutrients must be added to the Hyltebruk WWTP. Ideally, all dosed nutrients end up in the synthesised biomass. Therefore, the amount of nutrients required for appropriate treatment (including low levels of nutrients in the effluent) is closely linked to 1) The influent organic load, 2) The amount of biomass it yields and 3) The nutrient contents of this biomass. The conclusions related to these three factors are summarised below.

1) COD analyses, measuring the influent organic load, can be carried out automatically and relatively fast (1-2 hours) but does not always give a relevant picture of the pollutant content from the microorganisms point of view. Therefore, the COD tests often are complemented with respirometry or the more time consuming BOD test. From a nutrient dosage control perspective, BOD is not a suitable measurement variable as the results are delayed with at least five days from the sample occasion. Instead, control based on the organic load might depend on empirical correlations between COD and BOD. If the influent organic load does not show significant variations in its constitution, such a correlation is possible to find. It is established that COD:BOD correlations should be derived from BOD-tests carried out with biomass from the investigated WWTP.

2) Due to temperature-dependent decay processes, the observed yield, i.e. the parameter correlating the organic load with the sludge production, does not equal the theoretical yield. It is lowered by higher SRTs and increasing temperature. Moreover, at Hyltebruk the observed yield is significantly lowered by the BAS process design and the presence of higher-order organisms.

3) Pulp and paper activated sludges usually have lower nitrogen and phosphorus contents than those of municipal WWTPs. This is due to the generally nutrient deficient environments in which they are produced. The nutrient content of sludge is not constant but varies with the availability of soluble excess nitrogen and phosphorus. Therefore, nutrient dosage should be carried out with concurrent nutrient mass-balance analyses.

A well-working nutrient dosage control system generates several benefits such as reductions in nutrient discharges, fewer process disturbances and lower costs. However, the long time constants of biological systems and variable nutrient uptake, implies that over- or under-dosage is not always shown in residual nutrient concentrations. This makes the implementation of feedback control more difficult and a dynamic mathematical model is therefore an important tool for control system design. Several mathematical models are available describing COD removal processes, for example the Activated Sludge Model No. 1 (ASM1) (Henze *et al.*, 2000). They have, however, primarily been adapted to municipal wastewater treatment (WWT) characterised by relatively high concentrations of nitrogen and phosphorus. Consequently, they do not adequately

describe rate-limiting effects of nitrogen and phosphorus on microbial activity. In this work, the ASM1 therefore is extended to include:

- a proposed wastewater fractionation for phosphorus including five phosphorus state variables;
- limiting effects of ammonia nitrogen and soluble phosphorus on heterotrophic growth;
- nutrient regeneration through predation of higher-order organisms (protozoa) on bacteria;
- variable phosphorus uptake by active biomass depending on the available amount of exogenous soluble phosphorus.

In the proposed extended model, including protozoa, the influence of the sludge age or sludge retention time (SRT) on sludge production becomes apparent. At higher SRTs, the operational conditions allow the higher-order organisms to grow and compete with the heterotrophs, while at lower SRTs they tend to be washed out. The appearance of protozoa implies great challenges with regard to nutrient modelling. During energy transfer from bacteria to protozoa, energy is lost (from the system) due to inefficient biomass conversion. The nitrogen and phosphorus parts of the heterotrophic biomass not used for synthesis of protozoan biomass will be regenerated to the water phase and eventually be available for growth of bacteria.

The additional phosphorus uptake, or 'luxury uptake', process increases the time required for changes in influent soluble phosphorus concentration to propagate through the system and affect the effluent, i.e. it increases the time constant. Indications of such phenomena at Hyltebruk have been established. The presented model is validated for steady-state conditions and throughout Chapter 8, dynamic simulations demonstrate how the protozoa, the SRT and the concentration-dependent phosphorus uptake process influence nutrient requirements and the performance of the treatment process.

The specific two-stage design of the WWT plant presently in use at Hyltebruk, with production of dispersed bacteria in a biofilm (suspended carriers) stage, and subsequent consumption of these bacteria by protozoa in an activated sludge stage, reduces the total sludge production and complicates nutrient dosage. The total observed yield for the overall Hyltebruk WWTP is approximately 0.13 kg COD sludge produced/kg soluble COD reduced. In total, the plant removes 34 000 kg soluble COD per day and the mean nutrient content of the produced sludge is low, 4% N and 0.4% P (on a COD basis) throughout the entire plant. Consequently, the WWTP influent (phosphorus from the mill effluent + dosed phosphorus) should bring approximately 18 kg soluble bio-available phosphorus/d. However, BS reduces 45% of the influent soluble COD and the yield over this stage is 0.35 kg COD sludge produced/kg soluble COD reduced. If this is to be maintained, 25.2 kg/d of soluble phosphorus must be available in the influent. The conclusion is that the P requirements per day of the BS are 7.2 kg higher than for the plant in total. This corresponds to a soluble concentration of approximately 0.4 mg soluble P that theoretically will be found in the effluent if the plant is operated as during the measurement campaign.

This phenomenon could possibly be neutralised by lowering the sludge age so that the total sludge production equals the sludge production in the BS. A drawback with a decreased sludge

age is that the sludge production increases. Furthermore, instead of decreasing the P-requirements for the BS from 25 to 18 kg P/d, this approach increases the P-requirements of the overall plant from 18 to 25 kg P/d. A second solution would be to decrease the P-requirements of the BS. It has been reported that this is possible but that it may involve production of slime-forming bacteria in the BS. These organisms then seem to propagate unaffected throughout the AS and cause increased TSS concentrations in the effluent.

From the literature survey, it is clear that there are limits for residual concentrations of nutrients, below which the sludge composition starts to change and may result in an excess growth of filamentous organisms and/or poor reduction of COD. To complete a nutrient dosage control strategy, this limit must be determined. Possibly, 0.4 mg soluble P/l in the effluent is a reasonable set point for the Hylte WWTP. Then, if nutrient levels in the BS are maintained low, the plant can be operated as present.

## 9.1 Future work

Several types of problems and questions that deserve future attention have been encountered during this work:

- a strategy for model parameter estimation should be developed;
- further model validation with dynamic data is desirable;
- the usefulness of modelling the dispersed bacteria with a third independent active biomass state variable needs to be investigated;
- the usefulness of the double Monod-function modelling phosphorus limitations on heterotrophic growth should be evaluated;
- the additional phosphorus uptake process can be modelled so that the distributions of organisms with differing P-contents are considered;
- the model can be extended so that the nutrient content of higher-order organisms can differ from the content of floc-forming biomass;
- the availability of phosphorus assimilated in particulate material in the biofilm stage should be investigated;
- a more detailed analysis of the total sludge outtake at Hyltebruk would be valuable.

Some of these issues are strictly associated with model development, whereas some require extensive experimental laboratory work.



## Appendix

### Appendix A

#### A. Notation and abbreviations

AS	The activated sludge stage at Hylte
ASM1	The Activated Sludge Model No. 1
BAS	Combined biofilm (suspended carriers)/activated sludge process design
BOD	Biochemical oxygen demand
BOD <sub>∞</sub>	Ultimate biochemical oxygen demand
BPR	Biological phosphorus removal
BS	The suspended carrier biofilm stage at Hylte
$b$	General decay coefficient
$b_H$	Decay rate coefficient for heterotrophic organisms based on the death-regeneration hypothesis
$b_M$	Decay rate coefficient for protozoan organisms based on the death-regeneration hypothesis
C	Carbon
$C$	Concentration of total (soluble and particulate) matter
$C_{\text{COD}}$	Concentration of total (soluble and particulate) COD
$C_{\text{NTOT}}$	Concentration of total (soluble and particulate) nitrogen
$C_{\text{PTOT}}$	Concentration of total (soluble and particulate) phosphorus
COD	Chemical oxygen demand
CSTR	Continuously-stirred tank reactor
DOC	Dissolved organic carbon
d	Day
EBPR	Enhanced biological phosphorus removal
F/M	Ratio of food to biomass
$f$	General function
$f_P$	Fraction of heterotrophic biomass yielding (inert) particulate products based on the death-regeneration hypothesis
$f_{PM}$	Fraction of protozoa biomass yielding (inert) particulate products based on the death-regeneration hypothesis
$G_{ij}$	Mass flow of component $i$ in stream $j$
HRT <sub>aer</sub>	Aerobic hydraulic retention time
IWA	International Water Association
$i_{\text{col}}$	Soluble fraction of slowly degradable substrate
$i_{\text{XB}}$	Mass N/mass COD in active biomass
$\bar{i}_{\text{X,COD},n}$	The mean content of nutrient $n$ (N or P), in the different particulate components
$i_{\text{XIP}}$	Mass P/mass COD in inert particulate matter
$i_{\text{XP}}$	Mass N/mass COD in products from biomass decay
$i_{\text{XPP}}$	Mass P/mass COD in products from biomass decay
$i_{\text{XBP},1}$	Minimum mass P/mass COD in active biomass
$\hat{i}_{\text{XBP},2}$	Maximum additional mass P/mass COD in active biomass

$i_{XI}$	Mass N/mass COD in inert particulate matter
$K_{XBH}$	Half-saturation coefficient for consumption of heterotrophs by protozoa
$K_{O,H}$	Oxygen half-saturation coefficient for heterotrophs
$K_{O,M}$	Oxygen half-saturation coefficient for protozoa
$K_S$	Half-saturation coefficient for heterotrophic organisms
$K_X$	Half-saturation coefficient for hydrolysis of slowly biodegradable substrate
$k_a$	Ammonification rate
$k_h$	Maximum specific hydrolysis rate
MLSS	Mixed-liquor suspended solids
N	Nitrogen
$Q$	Local volumetric flow rate
$Q_{in}$	Influent volumetric flow rate to the WWT plant
$Q_r$	Recycled volumetric flow rate from the secondary settler underflow to the activated sludge basin influent
$Q_w$	Wastage volumetric flow rate from the settler underflow
$Q_{w,1}$	Wastage volumetric flow rate from the secondary settler underflow
$Q_{w,2}$	Wastage volumetric flow rate from the additional settler underflow
P	Phosphorus
$\mathbf{r}$	Reaction rate vector including net production rates per unit time and unit volume
$\mathbf{S}$	Stoichiometric matrix
$S$	Concentration of soluble material
$SP_{trad}$	The traditional definition of sludge production
$SP$	The definition of sludge production used throughout this work
SRT	Sludge age or sludge retention time
$S_{COD}$	Concentration of soluble COD
$S_I$	Concentration of soluble inert organic matter
$S_{ND}$	Concentration of soluble biodegradable nitrogen
$S_{NI}$	Concentration of soluble inert nitrogen
$S_{NH}$	Concentration of $NH_4^+ + NH_3$ nitrogen
$S_{NTOT}$	Concentration of soluble nitrogen
$S_O$	Concentration of oxygen
$S_p$	Concentration of soluble phosphorus available for assimilation
$S_{PO4}$	Concentration of ortho-phosphate
$S_{PTOT}$	Concentration of soluble phosphorus
$S_S$	Concentration of readily biodegradable substrate
T	Temperature
TKN	Total Kjeldahl nitrogen
TOC	Total organic carbon
TSS	Total suspended solids
$t$	Time
$V_{aer}$	Aerobic volume
VSS	Volatile suspended solids
WWT	Wastewater treatment
WWTP	Wastewater treatment plant
$X$	Concentration of particulate material
$X_{BM}$	Concentration of active protozoa biomass

$X_{\text{COD}}$	Concentration of particulate COD
$X_{\text{I}}$	Concentration of particulate inert organic matter
$X_{\text{H}}$	Concentration of active heterotrophic biomass
$X_{\text{NB}}$	Concentration of active mass nitrogen
$X_{\text{ND}}$	Concentration of particulate biodegradable nitrogen
$X_{\text{NI}}$	Concentration of particulate inert nitrogen
$X_{\text{NP}}$	Concentration of nitrogen in products arising from biomass decay
$X_{\text{NTOT}}$	Concentration of particulate nitrogen
$X_{\text{P}}$	Concentration of particulate products arising from biomass decay
$X_{\text{PB}}$	Concentration of total phosphorus in active biomass
$X_{\text{PB},1}$	Concentration of minimum phosphorus in active biomass
$X_{\text{PB},2}$	Concentration of additional phosphorus in active biomass
$X_{\text{PD}}$	Concentration of particulate biodegradable phosphorus
$X_{\text{PP}}$	Concentration of phosphorus in products arising from biomass decay
$X_{\text{PTOT}}$	Concentration of particulate phosphorus
$X_{\text{S}}$	Concentration of slowly biodegradable substrate
$Y$	True or theoretical carbonaceous yield constant
$Y_{\text{H}}$	True or theoretical carbonaceous yield coefficient for heterotrophs
$Y_{\text{M}}$	True or theoretical carbonaceous yield coefficient for protozoa
$Y^{\text{obs}}$	Observed carbonaceous yield coefficient

### Greek letters

$\alpha$	Weight coefficient for the double Monod function describing $S_{\text{p}}$ limitations on heterotrophic growth
$\xi$	Vector containing the model state variables
$\mu$	Growth rate function
$\hat{\mu}_{\text{H}}$	Maximum specific growth rate for heterotrophic organisms
$\hat{\mu}_{\text{M}}$	Maximum specific growth rate for protozoan organisms

### Subscripts

in	Wastewater stream entering the BS. In Chapter 8 sometimes wastewater entering the AS stage
$\text{bs}_i$	Wastewater in the two totally mixed suspended carrier biofilm reactors, line $i$
$\text{as}_i$	Wastewater in the two activated sludge basin effluents, line $i$
$\text{as}_{i,\text{sed}}$	Wastewater in the two secondary sedimentation unit effluents, line $i$
$w_{1,i}$	Wastewater in the two secondary settler underflows, line $i$
$w_2$	Wastewater in the additional settler underflow
eff	The WWT plant effluent

### Superscripts

obs	Observed
T	Transpose of a matrix

## Appendix B

### B. Parameters used in the extended model

Extended model parameters	Symbol	Unit	Used value	ASM1/ASM2d (30°C)
<i>Stoichiometric coefficients</i>				
Protozoan yield	$Y_M$	g cell COD formed (g cell COD oxidized) <sup>-1</sup>	0.335	-/-
Heterotrophic yield	$Y_H$	g cell COD formed (g COD oxidized) <sup>-1</sup>	0.67	0.67/0.625
Fraction of protozoan biomass yielding particulate products	$f_{PM}$	dimensionless	0.08	-/-
Fraction of heterotrophic biomass yielding particulate products	$f_P$	dimensionless	0.08	0.08/0.10
Nitrogen:				
Mass N/mass COD in active biomass	$i_{XB}$	g N (g COD) <sup>-1</sup>	0.05	0.086/0.07
Mass N/mass COD in products from biomass decay	$i_{XP}$	g N (g COD) <sup>-1</sup>	0.035	0.06/0.02
Mass N/mass COD in inert particulate matter	$i_{XI}$	g N (g COD) <sup>-1</sup>	0.04	-/0.02
Phosphorus:				
Minimum mass P/mass COD in active biomass	$i_{XBP,1}$	g P (g COD) <sup>-1</sup>	0.005	-/0.02 <sup>1</sup>
Mass P/mass COD in products from biomass decay	$i_{XPP}$	g P (g COD) <sup>-1</sup>	0.004	-/0.01
Mass P/mass COD in inert particulate matter	$i_{XIP}$	g P (g COD) <sup>-1</sup>	0.004	-/0.01
<i>Kinetic parameters</i>				
Protozoa:				
Protozoan max. specific growth rate	$\hat{\mu}_M$	day <sup>-1</sup>	1.2	-/-
Decay rate coefficient for protozoan organisms based on the death-regeneration hypothesis	$b_M$	day <sup>-1</sup>	0.1488	-/-
Half-saturation constant for heterotrophs	$K_{XBH}$	mg COD/l	7000	-/-

<sup>1</sup> This value is from ASM2d (Henze *et al.*, 2000) where the P content is assumed to be constant.

Half-saturation constant for oxygen	$K_{O,M}$	mg O <sub>2</sub> /l	0.2	-/-
Heterotrophs:				
Heterotrophic max. specific growth rate	$\hat{\mu}_H$	day <sup>-1</sup>	12	12/-
Decay rate coefficient for heterotrophic organisms based on the death-regeneration hypothesis	$b_H$	day <sup>-1</sup>	0.744	1.86/0.8
Half-saturation constant for oxygen	$K_{O,H}$	mg O <sub>2</sub> /l	0.2	0.2/0.2
Half-saturation constant for soluble substrate	$K_S$	mg COD/l	20	20/-
Maximum additional mass P/mass COD in active biomass	$\hat{i}_{XBP,2}$	g P (g COD) <sup>-1</sup>	0.005	-/-
Half-saturation constant for additional phosphorus uptake	$K_{P_i}$	mg P/l	0.2	-/-
Half-saturation constant for soluble phosphorus	$K_{P_1}$	mg P/l	0.01	-/- <sup>2</sup>
Half-saturation constant for soluble phosphorus	$K_{P_2}$	mg P/l	0.5	-/-
Half-saturation constant for ammonia nitrogen	$K_{NH}$	mg N/d	0.1	-/0.05 <sup>3</sup>
Ammonification rate	$k_a$	m <sup>3</sup> /(g COD day) <sup>-1</sup>	0.16	0.16/-
Max. specific hydrolysis rate	$k_h$	g slowly biodeg. COD (g cell COD day) <sup>-1</sup>	9.0	9.0/-
Half-saturation constant for hydrolysis of X <sub>s</sub>	$K_X$	g slowly biodeg. COD (g cell COD) <sup>-1</sup>	0.09	0.09/-

**Table B.1.** Parameters used in the extended model.

<sup>2</sup> ASM2d (Henze *et al.*, 2000) suggests a single Monod-function with  $K_P = 0.01$ . Alexandersson *et al.* (2003b) suggests that  $K_P$  (for a single Monod-function) is not higher than 0.05.

<sup>3</sup> ASM2d (Henze *et al.*, 2000) suggests that  $K_{NH} = 0.05$ . Alexandersson *et al.* (2003b) suggests that  $K_{NH}$  is not higher than 0.1.

## Appendix C

### C. Mean values from the measurement campaign

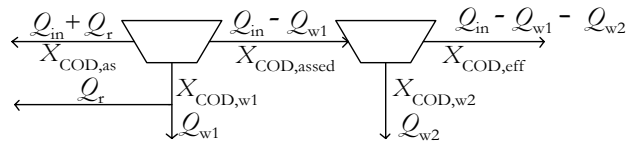
		in	bs	as			aS <sub>sed</sub>	eff
TSS	mg TSS/l	243 (33)	484 (127)	5387 (385)			18 (5)	4.1 (0.9)
VSS	mg VSS/l	164 (27)	377 (95)	4121 (254)			14 (4)	3.5 (0.4)
S <sub>COD</sub>	mg COD/l	2195 (192)	1233 (164)	329 (21)				
X <sub>COD</sub>	mg COD/l	317 (112)	656 (188)	6268 (998)				
S <sub>NTOT</sub>	mg N/l	5.7 (0.8)	6.7 (2.0)	3.9 (1.1)	C <sub>NTOT</sub>	mg N/l	5.4 (0.9)	5.2 (0.9)
X <sub>NTOT</sub>	mg N/l	9.0 (4.0)	23 (7.8)	247 (29)				
S <sub>NH</sub>	mg N/l	0.10 (0.00)	0.10 (0.00)	0.58 (0.68)	S <sub>NH</sub>	mg N/l	0.60 (0.47)	0.91 (0.44)
S <sub>PTOT</sub>	mg P/l	0.77 (0.16)	0.40 (0.12)	0.23 (0.10)	C <sub>PTOT</sub>	mg P/l	0.57 (0.13)	0.52 (0.11)
X <sub>PTOT</sub>	mg P/l	1.39 (0.31)	2.35 (0.25)	24.2 (3.2)				
S <sub>PO4</sub>	mg P/l	0.37 (0.14)	0.15 (0.07)	0.15 (0.08)	S <sub>PO4</sub>	mg P/l	0.27 (0.08)	0.36 (0.09)

**Table C.1.** Mean values from the measurement campaign (both lines combined). Standard deviations given in italic.

## Appendix D

### D. Estimation of particulate COD concentrations in the sludge flows

In Equation D.1 below, a steady-state expression for the return sludge particulate COD concentration is given. For notations, see Figure D.1. As  $X_{\text{COD,assed}} \approx 0$  and  $Q_{w1} \ll Q_r$ , we get that  $X_{\text{COD,w1}} \approx (1 + Q_{\text{in}}/Q_r) \cdot X_{\text{COD,as}}$ . Here, this means that the return sludge particulate COD concentration approximately should be twice that of the activated sludge particulate COD concentration. The analysis results did not support this.



**Figure D.1.** Schematic description of the notations in Appendix D.

In order to obtain reasonable values of the sludge concentrations in flows  $w_1$  and  $w_2$ , these were calculated from a mass balance over the settlers. It is assumed that the total inflow and total sludge outtake are split equally between the two lines. The concentrations may however differ, so the calculations have been made for each line separately.

$$(Q_{\text{in}} + Q_r)X_{\text{COD,as}} = (Q_{\text{in}} - Q_{w1})X_{\text{COD,assed}} + (Q_r + Q_{w1})X_{\text{COD,w1}} \quad (\text{D.1})$$

$$X_{\text{COD,w1}} = \frac{(Q_{\text{in}} + Q_r)X_{\text{COD,as}} - (Q_{\text{in}} - Q_{w1})X_{\text{COD,assed}}}{Q_r + Q_{w1}} \quad (\text{D.2})$$

No measurements of particulate COD in the settler effluent were made. However, this amount can be estimated using the  $X_{\text{COD}}/\text{TSS}$  ratios, which were approximately 1.2 in the AS units (see Appendix E). The same calculations were made for the additional sedimentation units:

$$(Q_{\text{in}} - Q_{w1})X_{\text{COD,assed}} = (Q_{\text{in}} - Q_{w1} - Q_{w2})X_{\text{COD,eff}} + Q_{w2}X_{\text{COD,w2}} \quad (\text{D.3})$$

$$X_{\text{COD,w2}} = \frac{(Q_{\text{in}} - Q_{w1})X_{\text{COD,assed}} - (Q_{\text{in}} - Q_{w1} - Q_{w2})X_{\text{COD,eff}}}{Q_{w2}} \quad (\text{D.4})$$

The results are shown in Table D.1. In the table,  $w_{1,i}$  denotes the sludge concentration in the secondary settler underflow, line  $i$ .  $w_{2,i}$  denotes the sludge concentration in the additional settler underflow, line  $i$ .

	$w_{1,1}$	$w_{1,2}$	$w_{2,1}$	$w_{2,2}$
$X_{\text{COD}}$ mg COD/l	11930	11800	885	1200
	(1350)	(2300)	(570)	(74)

**Table D.1.** Estimated concentrations of particulate COD in the sludge streams. Standard deviations are given in italic.

## Appendix E

### E. Ratios calculated from the measurement campaign

	in	bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1</sub>	as <sub>2</sub>	as <sub>1, sed</sub>	as <sub>2, sed</sub>	w <sub>1,1</sub>	w <sub>1,2</sub>	w <sub>2</sub>	eff
TSS/VSS	1.4 <i>(0.1)</i>	1.3 <i>(0.0)</i>	1.3 <i>(0.0)</i>	1.3 <i>(0.0)</i>	1.3 <i>(0.0)</i>	1.3 <i>(0.1)</i>	1.3 <i>(0.1)</i>	1.4 <i>(0.0)</i>	1.4 <i>(0.0)</i>	1.3	1.2 <i>(0.2)</i>
X <sub>COD</sub> /TSS	1.3 <i>(0.4)</i>	1.5 <i>(0.1)</i>	1.7 <i>(0.4)</i>	1.2 <i>(0.5)</i>	1.1 <i>(0.3)</i>						
X <sub>COD</sub> /VSS	1.8 <i>(0.65)</i>	1.9 <i>(0.14)</i>	2.1 <i>(0.5)</i>	1.6 <i>(0.2)</i>	1.5 <i>(0.3)</i>						
X <sub>NTOT</sub> /X <sub>COD</sub>	0.027 <i>(0.005)</i>	0.042 <i>(0.023)</i>	0.031 <i>(0.013)</i>	0.038 <i>(0.003)</i>	0.042 <i>(0.008)</i>						
X <sub>PTOT</sub> /X <sub>COD</sub>	0.0046 <i>(0.0008)</i>	0.0038 <i>(0.0014)</i>	0.0038 <i>(0.0008)</i>	0.0039 <i>(0.0011)</i>	0.0040 <i>(0.0007)</i>						

**Table E.1.** Various ratios calculated using measurements from the three day measurement campaign. Standard deviations given in italic.



## Appendix F

### F. Results from steady-state calculations

The results from the measurement campaign are used to calculate the flows of COD throughout the WWTP. The calculations are done according to the equations presented in Section 7.2.1. The results are presented in Table F.1.

		in	bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1, sed</sub>	as <sub>2, sed</sub>	w <sub>1,1</sub>	w <sub>1,2</sub>	w <sub>2,1</sub>	w <sub>2,2</sub>	eff
$G_{C,COD}$	ton COD/d	46 (7.7)	17 (3.7)	18 (3.9)	3 (0.3)	2.7 (0.7)	5.2*	4.8*	0.125*	0.169*	5.6 (0.5)
$G_{X,COD}$	ton COD/d	5.8 (2.4)	6.7 (2.5)	5.4 (1.6)							
$G_{S,COD}$	ton COD/d	40 (6.0)	10 (1.3)	12 (2.3)							

**Table F.1.** Total (C), particulate (X) and soluble (S) mass flows of COD along the Hylte WWTP. Mean values from the measurement campaign. Standard deviations are given in italic. \*Denotes that the values have been calculated from estimated sludge concentrations (Appendix D). Standard deviations given in italic.

With knowledge of the COD flows, COD reductions and the observed yields are calculated:

		bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1</sub>	as <sub>2</sub>	Overall plant
Reduction of soluble COD	%	25	20	18	23	86
Reduction of total COD	%	14	12	31	33	88
Observed yield coefficient	kg $X_{COD}$ / kg $S_{COD}$	0.36	0.31	-0.19*	-0.05*	0.13*

**Table F.2.** COD reductions (as % of total influent COD) and observed yield coefficients for the Hylte WWTP. \*Denotes that the values have been calculated from estimated sludge concentrations (Appendix D).

Sludge productions are calculated on the one hand directly from the campaign results and on the other with estimated sludge concentrations (Appendix D) according to the equations in Section 7.2.2

		bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1</sub>	as <sub>2</sub>	Overall plant
Sludge production	(ton/d)	3.8 (2.3)	2.5 (1.6)	-1.4* (2.9)	-0.4* (2.3)	4.5* (3.1)

**Table F.3.** Evaluation of sludge production. \*Denotes that the values have been calculated from estimated sludge concentrations (Appendix D).

The mass flows of phosphorus throughout the WWTP are calculated with the mass balance equations given in section 7.2. The results are listed in Table F.4

		in	bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1, sed</sub>	as <sub>2, sed</sub>	w <sub>1,1</sub>	w <sub>1,2</sub>	w <sub>2,1</sub>	w <sub>2,2</sub>	eff
$G_{C,PTOT}$	(kg P/d)	39 <i>(4.4)</i>	27 <i>(2.4)</i>	23 <i>(2.9)</i>	4.5 <i>(1.3)</i>	5.5 <i>(1.4)</i>	20* <i>(3.6)</i>	21* <i>(3.6)</i>	0.51* <i>(0.24)</i>	0.74* <i>(0.1)</i>	8.8 <i>(2.1)</i>
$G_{X,PTOT}$	(kg P/d)	25 <i>(6.4)</i>	23 <i>(2.8)</i>	20 <i>(2.5)</i>							
$G_{S,PTOT}$	(kg P/d)	14 <i>(2.5)</i>	3.7 <i>(1.5)</i>	3.5 <i>(0.5)</i>							

**Table F.4.** Total ( $C$ ), particulate ( $X$ ) and soluble ( $S$ ) mass flows of total P along the Hylte WWT. Mean values from the measurement campaign. Standard deviations given in italic. \*Denotes that the P-flows have been calculated using results from Appendix D and the ratio of P in particulate material (Appendix E).

## Appendix G

### G. Measurement campaign protocols

#### ANALYSER HYLTE 2003-05-20

Parameter	Unit	in	bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1</sub>	as <sub>2</sub>	as <sub>1, sed</sub>	as <sub>2, sed</sub>	eff	w <sub>1,1</sub> /w <sub>1,2</sub>	w <sub>2</sub>
$Q_{in}$	m <sup>3</sup> /d	17573	8787	8787	8770	8804	10554 ( $Q_t$ )	10554 ( $Q_r$ )	16115	865	281
Temperature	°C	34					36	36	33		
PH		7,3 (7.0)	7,8	7,8	8,2	8,3	8.2	8.1	8,2		
$C_{COD}/S_{COD}$	mg COD/l	2300/2000	1600/1100	1700/1200	7900/320	6400/320	330	330	320		
BOD <sub>7</sub>	mg BOD/l										
DOC	mg C/l	750	400	420	200	160	180	200	150		
TSS	mg TSS/l	280			5700	5000	13	20	3,2		
VSS	mg VSS/l				4400	3800	11	15	3,2		
SV <sub>30</sub>	ml/g TSS				140	120					
$S_O$	mg O <sub>2</sub> /l		2,8	3,8	2,5/1,5	2,0/1,5					
Aeration	m <sup>3</sup> /h		12300	12150	13530	15960					
$C_{NTOT}/S_{NTOT}$	mg N/l	16/5,1	42/9,8	24/5,2	280/5,2	220/3,7	6,2	5,7	5,5	280/660	96
$S_{NH}$	mg N/l	< 0,1	< 0,1	< 0,1	1,2	0,23	1,3	0,37	1,2		
$C_{PTOT}/S_{PTOT}$	mg P/l	2,3/0,67	3,2/0,62	2,5/0,34	24/0,31	22/0,36	0,58	0,64	0,62	23/52	8,6
$S_{PO4}$	mg P/l	0,42	0,11	0,13	0,19	0,29	0,32	0,36	0,44		

Dosage NP 26/6: 310 kg/d → 80,6 kg N/d, 18,6 kg P/d

Dosage Urea: 490 kg/d → 225,4 kg N/d

**ANALYSER HYLTE 2003-05-21**

Parameter	Unit	in	bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1</sub>	as <sub>2</sub>	as <sub>1, sed</sub>	as <sub>2, sed</sub>	eff	w <sub>1,1</sub> /w <sub>1,2</sub>	w <sub>2</sub>
$Q_{in}$	m <sup>3</sup> /d	19678	9839	9839	9837	9840	10511( $Q_r$ )	10511( $Q_t$ )	18196	865	281
Temperature	°C	35					35	35	33		
PH		7,1 (6.8)	7,8	7,7	8,3	8,3	8.2	8.1	8,2		
$C_{COD}/S_{COD}$	mg COD/l	2800/2400	2100/1190	2200/1500	6600/340	5200/340	360	360	330		
BOD <sub>7</sub>	mg BOD/l	1300/1200	320	560	< 8	< 8	13	11	9		
DOC	mg C/l	820	420	500	142	170	140	160	160		
TSS	mg TSS/l	240	660	380	5300	5900	24	19	5,1	4800/8300	2200
VSS	mg VSS/l	188	510	297	4000	4500	18	15	4	3600/6300	1700
SV <sub>30</sub>	ml/g TSS										
$S_O$	mg O <sub>2</sub> /l		2.4	3.9	2.4/1.5	1.5/1.5					
Aeration	m <sup>3</sup> /h		11800	11900	14700	16400					
$C_{NTOT}/S_{NTOT}$	mg N/l	17/5,4	33/6,1	21/18	230/4,7	250/3,9	6,2	5,6	6	240/360	110
$S_{NH}$	mg N/l	< 0,1	0,1	< 0,1	1,7	0,16	1,2	0,3	1,1		
$C_{PTOT}/S_{PTOT}$	mg P/l	2,2/0,68	2,9/0,25	2,7/0,41	21/0,15	23/0,27	0,59	0,74	0,54	22/33	11
$S_{PO4}$	mg P/l	0,48	0,23	0,26	0,08	0,18	0,26	0,33	0,39		

Dosage NP 26/6: 360 kg/d → 93,6 kg N/d, 21,6 kg P/d

Dosage Urea: 510 kg/d → 234,6 kg N/d

## ANALYSER HYLTE 2003-05-22

Parameter	Unit	in	bs <sub>1</sub>	bs <sub>2</sub>	as <sub>1</sub>	as <sub>2</sub>	as <sub>1, sed</sub>	as <sub>2, sed</sub>	eff	(w <sub>1,1</sub> /w <sub>1,2</sub> )	w <sub>2</sub>
$Q_{in}$	m <sup>3</sup> /d	17324	8662	8662	8679	8645	8671 ( $Q_r$ )	8671 ( $Q_r$ )	15793	865	282
Temperature	°C	35					35	35	33		
PH		7,1 (6,9)	7,7	7,7	8,3	8,3	8,1	8,1	8,2		
$C_{COD}/S_{COD}$	mg COD/l	2400/2200	1900/1100	1900/1300	6000/320	7400/310	340	340	340		
BOD <sub>7</sub>	mg BOD/l										
DOC	mg C/l	650	320	410	120	110	110	120	110		
TSS	mg TSS/l	215	490	410	5000	5500	10	22	4	8300/6000	75 (?)
VSS	mg VSS/l	145	380	325	4930	4200	7,6	18,4	3,2	6400/4600	62(?)
SV <sub>30</sub>	ml/g TSS				120	120					
$S_O$	mg O <sub>2</sub> /l		3,0	3,5	2,5/1,5	2,0/1,5					
Aeration	m <sup>3</sup> /h		11680	11860	12713	15716					
$C_{NTOT}/S_{NTOT}$	mg N/l	11/6,6	30/7,3	17/5	240/2,9	280/2,8	4	4,9	4,2	380/280	13
$S_{NH}$	mg N/l	< 0,1	0,11	< 0,1	0,13	< 0,1	0,29	0,24	0,41		
NO <sub>3</sub> -N	mg N/l					< 0,05					
$C_{PTOT}/S_{PTOT}$	mg P/l	2/0,96	2,8/0,38	2,4/0,4	29/0,16	26/0,15	0,36	0,51	0,4	37/30	1,8
$S_{PO4}$	mg P/l	0,22	0,08	0,11	0,09	0,08	0,15	0,19	0,26		

AS1-1 COD = 7800/370 N = 270/5 NH<sub>4</sub>-N = <0,1 DOC = 140 P-tot = 29/0,12  
AS1-2 COD = 7200/330 N = 240/3,3 NH<sub>4</sub>-N = 0,1 DOC = 120 P-tot = 27/0,13  
AS1-3 COD = 6200/320 N = 240/3,4 NH<sub>4</sub>-N = 0,16 DOC = 120 P-tot = 25/0,17

## Appendix H

### H. Influence of sludge age on the observed yield coefficient

Assume a biomass population  $X$ , growing on substrate  $S$  in a traditional AS process (Figure 2.1). Assume further that the system is in steady-state, i.e. the derivatives are zero and that the concentrations of particulate material in the influent and in the settler overflow are approximately zero:

$$V \cdot \frac{dS}{dt} = Q_{in} \cdot (S_{in} - S) - \frac{1}{Y} \cdot \mu \cdot X \cdot V = 0 \quad (H.1)$$

$$V \cdot \frac{dX}{dt} = -Q_w \cdot X_w - b \cdot X \cdot V + \mu \cdot X \cdot V = 0 \quad (H.2)$$

Here,  $\mu$  and  $b$  are growth and decay rates, respectively. The observed yield can be defined as:

$$Y^{obs} = \frac{G_x}{G_s} = \frac{\text{sludge production}}{\text{substrate consumption}} \quad (H.3)$$

According to H.1:

$$G_s = Q_{in} \cdot (S_{in} - S) = \frac{1}{Y} \cdot \mu \cdot X \cdot V \quad (H.4)$$

According to H.2:

$$G_x = Q_w \cdot X_w = \mu \cdot X \cdot V - b \cdot X \cdot V \quad (H.5)$$

H.4 together with H.5 gives:

$$G_x = Y \cdot G_s - b \cdot X \cdot V \quad (H.6)$$

If the suspended solids concentration in the settler effluent is assumed to be zero, the aerobic sludge age can be written:

$$SRT_{acr} = \frac{X \cdot V}{G_x} \quad (H.7)$$

$$\rightarrow G_x = Y \cdot G_s - b \cdot SRT_{acr} \cdot G_x \quad (H.8)$$

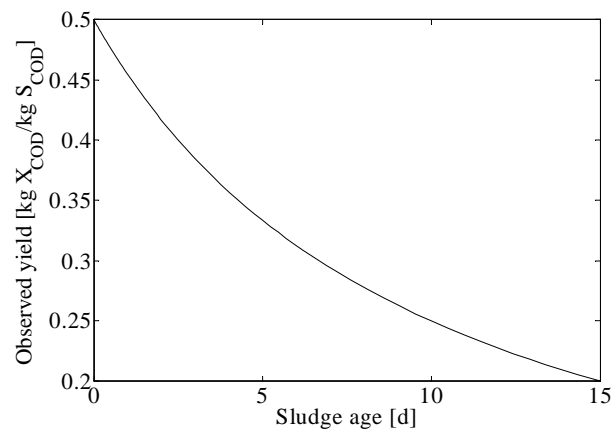
By combining Equations H.8 and H.3, we conclude:

$$1 = Y \underbrace{\frac{G_S}{G_X}}_{1/Y^{\text{obs}}} - b \cdot \text{SRT}_{\text{aer}} = \frac{Y}{Y^{\text{obs}}} - b \cdot \text{SRT}_{\text{aer}} \quad (\text{H.9})$$

and consequently:

$$Y^{\text{obs}} = \frac{Y}{1 + b \cdot \text{SRT}_{\text{aer}}} \quad (\text{H.10})$$

In Figure H.1, the observed carbonaceous yield coefficient are plotted as a function of the sludge age (or sludge retention time (SRT)).



**Figure H.1.** The observed carbonaceous yield according to Equation H.10. In this case,  $Y=0.5$  kg  $X_{\text{COD}}/\text{kg } S_{\text{COD}}$  and  $b=0.1 \text{ d}^{-1}$ .

## Bibliography

Alexandersson, T. (2003), *Water Reuse in Paper Mills*, Measurements and Control Problems in Biological Treatment. Licentiate thesis, ISBN 91-88934-28-4, Department of Industrial Electrical Engineering and Automation, Lund University, Lund, Sweden.

Alexandersson, T., Rosen, C., Jeppsson, U. (2003a), Framework for a control strategy of in-mill biological treatment using on-line sensors and dynamic modelling. *Wat. Sci. Tech.* Accepted for publication.

Alexandersson, T., Jeppsson, U., Rosen, C. (2003b), Biological treatment of whitewater in a laboratory process in order to determine kinetic parameters for model development. *Wat. Sci. Tech.* Accepted for publication.

Copp, J.B. (ed.) (2002), *The COST simulation benchmark - Description and simulator manual*, ISBN 92-894-1658-0, Office for Official Publications of the European Communities, Luxembourg.

Doran, P.M. (1995), *Bioprocess engineering principles*. Academic Press Limited, San Diego, California, England.

Eckenfelder, W.W.Jr. (1989), *Industrial Water Pollution Control*, 2<sup>nd</sup> ed., McGraw-Hill, cop., New York, USA.

Eckenfelder, W.W., Grau, P. (1992), *Activated sludge process design and control*. Technomic Publishing Company, Inc. Lancaster, Pennsylvania, USA.

Foster, H.F., Redmond, A., Crozier, S. (1999), Optimization of nutrient control in a paper mill activated sludge system. *TAPPI Proceedings, 1999 International Environmental Conference*, 711-715.

Grau, P. (1991), Criteria for nutrient balanced operation of activated sludge plants. *Wat. Sci. Tech.*, **24**(3-4), 251-258.

Henze, M., Harremoës, P., Jansen, J. la C., Arvin, E. (1995), *Wastewater treatment, biological and chemical processes*. Springer-Verlag, Berlin, Germany.

Henze, M., Gujer, W., Mino, T., van Loosdrecht, M. (2000), Activated sludge models ASM1, ASM2, ASM2d, and ASM3. *Scientific and Technical Report No. 9*. IWA Publishing, London, England.

Hynninen, P., Viljakainen, E. (1995), Nutrient dosage in biological treatment of wastewaters. *Tappi Journal*, **78**(5), 105-108.

Jansson, K., Järvinen, S., Koponen, R., Temmes, A. (1994), Control of nutrient discharges – practical experiences. *Wat. Sci. Tech.*, **29**(5-6), 343-351.



- Jeppsson, U. (1996), *Modelling aspects of wastewater treatment processes*. PhD thesis, ISBN 91-88934-00-4, Department of Industrial Electrical Engineering and Automation, Lund University, Lund, Sweden.
- Järvinen, R. (1997), Nitrogen in the effluent of the pulp and paper industry. *Wat. Sci. Tech.*, **35**(2-3), 139-145.
- Möbius, C.H. (1991), Nitrogen and phosphorus limits for nutrient deficient industrial wastewaters. *Wat. Sci. Tech.*, **23**(3-4), 259-267.
- Ning, Z., Gilles, G.P., Spanjers, H. (2000), Identification and quantification of nitrogen nutrient deficiency in the activated sludge process using respirometry. *Wat. Res.*, **34**(13), 3345-3354.
- Olsson, G., Newell, B. (1999), *Wastewater treatment systems: modelling, diagnosis and control*. IWA Publishing, London, England.
- Richard, G. R. (1999), Causes for variable phosphorus needs in pulp and paper activated sludge systems. *TAPPI Proceedings, 1999 International Environmental Conference*, 703-709.
- Saunamäki, R. (1994), Experimental study on the control of nutrients in activated sludge treatment. *Wat. Sci. Tech.*, **29**(5-6), 329-342.
- Takács, I., Patry, G.G., Nolasco, D. (1991), A dynamic model of the clarification-thickening process. *Wat. Res.*, **25**(10), 1263-1271.
- Thompson, G., Swain, J., Kay, M., Forster, C.F. (2001), The treatment of pulp and paper mill effluent: a review. *Bioresource Technology*, **77**, 275-286.
- Verduyn, C. (1991), *Energetic aspects of metabolic fluxes in yeasts*. PhD thesis, Delft University of Technology, Delft, the Netherlands.
- Weijers, S.R. (1999), On BOD tests for the determination of biodegradable COD for calibrating activated sludge model no.1. *Wat. Sci. Tech.*, **39**(4), 177-184.