Department of Civil and Environmental Engineering

The effect of flow equalization and low-rate prefermentation on the activated sludge process and biological nutrient removal

Anna Mikola

Aalto University

DOCTORAL DISSERTATIONS

Aalto University publication series
DOCTORAL DISSERTATIONS 76/2013

The effect of flow equalization and low-rate prefermentation on the activated sludge process and biological nutrient removal

Anna Mikola

A doctoral dissertation completed for the degree of Doctor of Science (Technology) to be defended, with the permission of the Aalto University School of Engineering, at a public examination held at the lecture hall R1 of the school on 24 May 2013 at 12.

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Aalto University publication series
DOCTORAL DISSERTATIONS 76/2013

ISBN 978-952-60-5152-9 (printed)
ISBN 978-952-60-5153-6 (pdf)
ISSN-L 1799-4934
ISSN 1799-4934 (printed)
ISSN 1799-4942 (pdf)
http://urn.fi/URN:ISBN:978-952-60-5153-6

Unigrafia Oy
Helsinki 2013

Finland

NORDIC ECOLABEL
441 697
Printed matter

Aalto University
Aalto University, P.O. Box 11000, FI-00076 Aalto www.aalto.fi

Abstract

Author
Anna Mikola

Name of the doctoral dissertation
The effect of flow equalization and low-rate prefermentation on the activated sludge process and biological nutrient removal

Publisher School of Engineering

Unit Department of Civil and Environmental Engineering

Series Aalto University publication series DOCTORAL DISSERTATIONS 76/2013

Field of research Water Engineering Research Group

Manuscript submitted 29 January 2013 **Date of the defence** 24 May 2013

Permission to publish granted (date) 11 March 2013 **Language** English

☐ **Monograph** ☒ **Article dissertation (summary + original articles)**

Abstract

The flow and load variations in the wastewater plant influent complicate the operation of the biological treatment process and harm the process performance. Moreover, when the biological nutrient removal (BNR) process is implemented in the plant, the plant influent is often lacking readily biodegradable organic matter. Readily biodegradable organic matter can be produced by prefermentation. In this research project, the primary clarifiers at the Pihlajaniemi WWTP in Savonlinna were modified in order to tackle the problems caused by flow variation and the lack of suitable organic matter.

This study demonstrated that diurnal flow variations were efficiently levelled out in the existing primary clarifier basin volume. Surprisingly, a significant amount of organic matter was transformed into a more accessible form for the BNR bacteria when only flow equalization was in operation, but the attempt at enhancing the VFA production by adding an internal sludge recycle was not successful. The raw sludge removal in the equalization/prefermentation basin was not compromised in the modified operation. There is no commonly accepted method for assessing the magnitude of flow variations or the efficiency of the flow equalization. This thesis introduces a coefficient of flow variation that can be used for this purpose.

The modifications in the pre-treatment were beneficial for the process performance. The main improvement in the biological process performance was observed with nitrification. The improvement could be mainly attributed to the diurnal flow equalization. Moreover, it could be demonstrated that the increased heterotrophic assimilation and also prefermentation influenced nitrification. The results on nitrification are of high importance because the effects of dynamic influent and feed water characteristics on nitrification have not been widely studied, especially not in full-scale. Moreover, the sludge settling characteristics were improved in the equalization/prefermentation process train compared with the reference process train. This, together with the more constant flow rate, enabled better hydraulic control of the secondary clarifiers.

It can be concluded that the modification of the existing primary clarifier to a multi-functional pre-treatment basin is a feasible solution for the improvement of the BNR process performance. This process modification could be widely implemented in Finland because the majority of middle-sized and large WWTPs have primary clarifiers. Nitrification is usually the limiting part of the biological wastewater treatment. Therefore, implementation of equalization/prefermentation would enable a reduction in the aerated process volume. The economic balance of the modifications is clearly positive.

Keywords Flow equalization, prefermentation, biological nutrient removal

ISBN (printed) 978-952-60-5152-9 **ISBN (pdf)** 978-952-60-5153-6

ISSN-L 1799-4934 **ISSN (printed)** 1799-4934 **ISSN (pdf)** 1799-4942

Location of publisher Espoo **Location of printing** Helsinki **Year** 2013

Pages 157 **urn** http://urn.fi/URN:ISBN:978-952-60-5153-6

Aalto-yliopisto

Aalto-yliopisto, PL 11000, 00076 Aalto www.aalto.fi

Tiivistelmä

Tekijä
Anna Mikola

Väitöskirjan nimi
Virtaaman tasauksen ja esifermennoinnin vaikutus aktiivilieteprosessiin ja biologiseen ravinteiden poistoon

Julkaisija Insinööritieteiden korkeakoulu

Yksikkö Yhdyskunta- ja ympäristötekniikan laitos

Sarja Aalto University publication series DOCTORAL DISSERTATIONS 76/2013

Tutkimusala Vesitekniikka

Käsikirjoituksen pvm 29.1.2013 **Väitöspäivä** 24.5.2013

Julkaisuluvan myöntämispäivä 11.3.2013 **Kieli** Englanti

☐ **Monografia** ☒ **Yhdistelmäväitöskirja (yhteenveto-osa + erillisartikkelit)**

Tiivistelmä

Jätevedenpuhdistamoiden tulevan veden virtaama- ja kuormitusvaihtelut hankaloittavat biologisen käsittelyn ohjausta ja huonontavat prosessin puhdistustehoa. Kun biologinen ravinteiden poisto otetaan käyttöön jätevedenpuhdistamolla, ei tuleva vesi useinkaan sisällä riittävästi helposti hajoavaa orgaanista ainetta. Helposti hajoavan orgaanisen aineen määrää voidaan kasvattaa esifermennoinnin avulla. Tässä tutkimusprojektissa esiselkeytysaltaita Savonlinnan Pihlajaniemen puhdistamolla muokattiin niin, että puhdistamo voisi paremmin selviytyä virtaamavaihtelun ja sopivan orgaanisen aineen puutteen aiheuttamista ongelmista.

Tämä tutkimus osoittaa, että virtaaman vuorokausivaihtelut voidaan tasata olemassa olevassa esiselkeytystilavuudessa. Yllättäen merkitsevä osa orgaanisesta aineesta muuttui bakteereille käytettävissä olevaan muotoon, kun allasta käytettiin vain virtaaman tasaukseen. Sisäinen lietteen kierrätyksen lisääminen altaaseen orgaanisen esifermennoinnin parantamiseksi ei tehostanut hydrolyysiä altaassa. Muokattu esikäsittelyn toiminta ei vaikeuttanut raakalietteen poistamista altaasta. Koska yleisesti hyväksyttyä menetelmää virtaamavaihteluiden suuruuden tai tasausaltaan tehokkuuden arvioimiseksi ei ole, tässä tutkimuksessa kehitettiin virtaaman vaihtelukerroin, jota voidaan käyttää tähän tarkoitukseen.

Muokattu esikäsittely paransi prosessin toimintaa. Suurin parannus biologisessa prosessissa havaittiin nitrifikaation toiminnassa, joka voitiin selittää pääasiassa virtaaman vuorokausivaihtelun tasaamisella. Lisäksi osoitettiin, että kasvanut heterotrofinen assimilaatio sekä esifermentointi vaikuttivat nitrifikaation toimintaan. Nitrifikaation toimintaa selittävät tulokset ovat tärkeitä, koska tulevan veden laadun ja kuormituksen vaihtelun vaikutusta ei ole laajasti tutkittu, ei varsinkaan täyden mittakaavan kokeissa. Lisäksi todettiin, että lietteen laskeutuvuus oli parempi tasatulla ja esifermentoidulla käsittelylinjalla verrattuna vertailulinjaan. Lietteen paremmat ominaisuudet sekä tasaantunut virtaama mahdollistivat myös sen, että jälkiselkeytysaltaiden hydraulinen hallinta onnistui paremmin.

Voidaan todeta, että olemassa olevien esiselkeytysaltaiden muokkaaminen monikäyttöisiksi esikäsittelyaltaiksi on toteutuskelpoinen ratkaisu, kun halutaan parantaa biologisen ravinteiden poistoprosessin toimintaa. Tämä voitaisiin toteuttaa laajasti Suomessa, koska suurimmalla osalla keskikokoisista ja suurista puhdistamoista on esiselkeytysaltaat. Nitrifikaatio on yleensä rajoittava osa biologisen jätevedenkäsittelyn toiminnassa. Näin ollen lisäämällä tasaus ja esifermentointi puhdistusprosessiin voitaisiin pienentää ilmastettua prosessitilavuutta.

Avainsanat virtaaman tasaus, esifermentointi, biologinen ravinteiden poisto

ISBN (painettu) 978-952-60-5152-9 **ISBN (pdf)** 978-952-60-5153-6

ISSN-L 1799-4934 **ISSN (painettu)** 1799-4934 **ISSN (pdf)** 1799-4942

Julkaisupaikka Espoo **Painopaikka** Helsinki **Vuosi** 2013

Sivumäärä 157 **urn** http://urn.fi/URN:ISBN:978-952-60-5153-6

Acknowledgements

The experimental part of this study was carried out at the Pihlajaniemi plant in the City of Savonlinna. The study was supervised by a group of Finnish wastewater specialists: Ari Niemelä, Saijariina Toivikko, Federico Varalta, Hannu Lalu, Jorma Räsänen, Jyri Rautiainen, Niko Rissanen, Pentti Karhu, Pirjo Rantanen and Riku Vahala. I am grateful to all the members of the group for valuable comments and critique during the experiments.

I wish to thank the people from Savonlinna Water and Pihlajaniemi WWTP for once again submitting to uncertainties of almost endless full-scale tests. I am grateful for all the staff for making the arrangements at the plant. I also enjoyed the lunch and coffee breaks. Special thanks for memorable discussions to Teijo Brunou, who took care of the control application of the equalization pump. Paavo Tahvanainen and Virpi Luukkanen are greatly acknowledged for taking care of most of the sampling and analyzing during the study. I am especially grateful to Paavo for sharing his long experience and insight on biological nutrient removal process during the study and later on.

I would like to express my gratitude to two professors who greatly contributed for the materialization of the research project and this thesis. Prof. Heikki Kiuru introduced the idea of the modified pre-treatment and saw the value of our results. Prof. Kiuru also provided me a homelike accommodation in his home town Savonlinna for almost each of my plant visits. Prof. Riku Vahala contributed to the start of the project at the beginning in another role and has given valuable advice, comments and encouragement during the latter part of the work. Both professors have supported and inspired me during the work.

I am very grateful to the people at the Water Engineering Research group. Thank you Anne-Mari Aurola for your help with the design of the analytical part; Ritva Laitala for your help with the RBCOD analysis; Henri Haimi and Michela Mulas for your valuable comments to a couple of my papers; Pirjo Rantanen for your great comments to my papers and manuscript; Ari Järvinen for your enormous help with all the practicalities throughout the project, for the management of the whole study and for your unique kindness; and finally Aino Peltola for conducting the VFA analyses, for the help with the analytical part and most importantly for your caring and enthusiasm.

I have been extremely lucky in meeting many great colleagues during my career so far- first at Nopon Oy, then at Kiuru&Rautiainen Oy and now Ramboll. I would like to thank Mari Heinonen and Tommi Fred for good discussions and their support.

Special thanks to Teemu Heikkinen for the help with maps and for many enjoyable talks. My colleagues from Savonlinna have helped me to discover the beautiful Savonlinna region. I would especially like to express my gratitude to Jyri Rautiainen for his unselfish support during the whole project and for his aspiration to increase the understanding of the water environment.

I am also most thankful to Dr. Julian Sandino who made me feel comfortable in the international wastewater research community.

Finally, I would like to express my gratitude to Prof. Jan A. Oleszkiewicz and to Prof. Jacek Makinia for pre-examining my manuscript and giving very useful comments for it. Prof. Karin Jönsson is greatly acknowledged for being the opponent for the dissertation.

For financial support of this study, I acknowledge the National Technology Agency of Finland (TEKES), Maa- ja vesitekniikan tuki ry, Finnish Water Utilities Association (FIWA), Savonlinna Water, ABS Nopon Ltd, Grundfos Pumps Ltd, Kiteen Vesikunta, Kiuru&Rautiainen Oy and TKK Foundation.

My deepest gratitude goes to my dear family: Gil, Fanny, Ellen, Mia and Emil, my mother Pirkko, mother-in-law Mireille, my brother Juha and sister Eeva. Your support and baby-sitting have been priceless.

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Abbreviations

ADN	Process with anaerobic, anoxic and aerobic sections in sequence
ADND	Process with anaerobic, anoxic, aerobic and anoxic sections in sequence
AND	Process with anaerobic, aerobic and anoxic sections in sequence
AOB	Ammonium oxidizing bacteria
A/O process	Process with anaerobic and aerobic sections in sequence
AOR	Actual oxygen requirement
APT	Activated primary treatment
BNR	Biological nutrient removal
BOD	Biological oxygen demand
BOD_7(ATU)	7-day biological oxygen demand with the addition of allylthiourea
C/N	Carbon to nitrogen ratio
C/P	Carbon to phosphorus ratio
COD	Chemical oxygen demand
COD_{Cr}	Chemical oxygen demand with oxidation using dichromate
DAF	Dissolved air flotation
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DSVI	Diluted sludge volume index
EBPR	Enhanced biological phosphorus removal
FTU	Formazin turbidity unit
GAO	Glycogen-accumulating organism
GHG	Greenhouse gas
HRT	Hydraulic retention time
HUT	Helsinki University of Technology
I/I	Infiltration/inflow
MLSS	Mixed-liquor suspended solids, sludge concentration in the aeration basin
NOB	Nitrite oxidizing bacteria
ORP	Oxydation-reduction potential
OUR	Oxygen uptake rate
PAO	Polyphosphate-accumulating organism
PE	Population equivalent
PHA	Polyhydroxyalkanoates
PHB	Polyhydroxybutyrate
Q	Flow rate
RAS	Return-activated sludge

RBCOD	Readily biodegradable COD
r_{VFA}	Rate of VFA production
S	Substrate
SBCOD	Slowly biodegradable COD
SER	Sludge elutriation rate
SFS	Finnish Standard Association
SRR	Sludge recycle rate
SRT	Sludge retention time (= sludge age)
SS	Suspended solids
SVI	Sludge volume index
TKN	Total Kjeldahl nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UCT	University of Cape Town, a process type
WAS	Waste-activated sludge
VFA	Volatile fatty acids
VSS	Volatile suspended solids
WWTP	Wastewater treatment plant
X	Sludge concentration
ΔC_{VFA}	VFA concentration increase in total wastewater flow due to prefermenter
θ	Temperature coefficient in Arrhenius equation

List of original publications

This thesis is based on the following five articles (appendices), which are referred to by their roman numerals throughout the text (e.g. 'Appendix I' or simply 'I').

I Mikola, A., Rautiainen, J., Kiuru H. (2007). "Diurnal flow equalization and prefermentation using primary clarifiers in a BNR plant." *Water Practice* **1** (5) 1-13

The author planned the study, carried out a part of the experimental work, analyzed the data and had the main responsibility for writing the paper.

II Mikola, A.M.K.,Rantanen P., Vahala, R., Rautiainen, J.A. (2012). "Full-scale investigation of the influence of flow equalization and prefermentation on nitrification." *Water Environ. Res.**84** (5) 452-459*

The author planned the study, carried out a part of the experimental work, analyzed the data and had the main responsibility for writing the paper.

III Mikola, A., Rautiainen, J., Vahala, R. (2009). "Secondary clarifier conditions conducting to secondary phosphorus release in a BNR plant." *Water Sci. Technol.- WST* **60** (9) 2413-2418

The author planned the study, carried out a part of the experimental work, analyzed the data from years 2001-2003 and had the main responsibility for writing the paper.

IV Mikola, A., Rautiainen, J., Kiuru H. (2008). "The Effect of Flow Equalization and Prefermentation on the Sludge Production and Sludge Characteristics in a BNR Plant." *Water Sci. Technol. – WST* **57** (12) 2023-2029

The author planned the study, carried out a part of the experimental work, analyzed the data and had the main responsibility for writing the paper.

V Mikola, A.M.K., Vahala, R., Rautiainen, J.A. (2011). "Factors affecting the quality of the plant influent and its suitability for prefermentation and the biological nutrient removal process." *J Environ Eng* **137** 1185

The author planned the study, carried out a part of the experimental work, analyzed the data and had the main responsibility for writing the paper.

1 Introduction

1.1 Background

This thesis focuses on two challenges of biological wastewater treatment caused by the characteristics of the influent wastewater – flow variations and the quality of the organic matter. The wastewater flow and load arriving in the wastewater treatment plant (WWTP) vary considerably. The variations have more or less regular diurnal, weekly and seasonal patterns. These variations have a detrimental effect on the biological treatment process. Therefore, equalization can be implemented in the process prior to the biological reactor. Equalization is used worldwide, but its popularity is slackened by the required basin volume. Furthermore, when biological nutrient removal (BNR) is introduced in the wastewater treatment plant, a sufficient amount of readily biodegradable organic matter is needed. Primary clarifiers in their traditional role of organic load removal often do not respond to the requirements of the BNR process. In this process, different groups of bacteria, called BNR bacteria, are responsible for nitrification, denitrification and enhanced biological phosphorus removal. Denitrification and phosphorus removal efficiencies are dependent on the presence of suitable organic carbon. Prefermentation is a process where suitable organic matter for the BNR bacteria is produced from wastewater's own carbon. This process has become popular in many countries but has not yet gained wide popularity in Finland.

A full-scale 18-month study was carried out at the Pihlajaniemi WWTP in the City of Savonlinna in Eastern Finland. The role of primary clarification had become less important at the Pihlajaniemi plant after the implementation of BNR. Moreover, during wintertime, the capacity of the plant was not sufficient to achieve full nitrogen removal. Therefore, the idea of modifying the existing primary clarifiers in the equalization basin seemed worth testing. Equalization has the potential for increasing the capacity of the biological process. Also, by using the existing basins, the problems and costs related to a new basin volume for equalization were solved. At times, a lack of organic matter was limiting enhanced biological phosphorus removal and denitrification at the plant. Thus, the primary clarifier was also used for prefermentation with the internal raw sludge recycle transferring the soluble organic matter from the sludge blanket to the liquid phase. In spite of these modifications, raw sludge was still removed in order to avoid conducting excessive load and inert material in the biological process.

1.2 Objectives of the study

The objective of the study was to test the pre-treatment process with equalization and prefermentation implemented in existing primary clarifiers and to answer the following questions:

1. How did the pre-treatment perform?
2. What was the effect on the biological treatment process?
3. What was the effect on sludge production and energy consumption?
4. How could the need for equalization and prefermentation be assessed in advance?

The first research question above is answered in Appendix I. The results concerning the effects of prefermentation on denitrification, on phosphorus release and on the hydraulic control of the plant are also presented in Appendix I. Nevertheless, important results were found on less researched effects of the flow equalization and prefermentation, namely effects on nitrification, on phosphorus removal and on sludge characteristics. Appendix II answers the second research question and focuses on nitrification. Appendix III deals further with this question and discusses the role of secondary clarification on phosphorus removal. Appendix IV answers the third research question concerning sludge production. Energy consumption is discussed separately in Section 3.4 of this thesis.

Along the work, it became evident that it would not be enough to understand the biological wastewater treatment process alone. Flow equalization is not solely achieved in the equalization basins and storage tanks. As a matter of fact, flow equalization starts in households and industry and it is influenced by the sewer network design and by the efforts to minimize infiltration/inflow (I/I). In addition, some flow equalization could be obtained in every treatment plant by the intelligent control of reject flows and by the addition of by-pass options. Thus, in order to consider whether an equalization or prefermentation basin would be useful and to successfully operate them, it is necessary to understand the integrated wastewater system. This gave rise to the fourth research question, which is partly answered in Appendix V. Moreover, this thesis contains a separate description of wastewater sources and transformations in the sewer system. Although the study did not include measurements from the sewer network, an attempt was made to understand the effect of the sewer network conditions on the quality and quantity of plant influent water.

1.3 Structure of the thesis

This thesis consists of four sections. In Materials and Methods (Chapter 2), the full-scale research project is described in detail, including the operation procedure chosen for the multifunctional pre-treatment basin and a description of sampling campaigns. The chapter also gives a comprehensive view of the Pihlajaniemi WWTP. Because significant efforts towards flow and load equalization and sufficient organic feed for BNR had been implemented at the plant already earlier, these measures are also described and a short assessment of their influence is given. Chapter 3 is divided into sections. Each section contains a literature review first, followed by findings from the Pihlajaniemi WWTP concerning the topic and finally a short discussion. This chapter follows the logic of the four research questions with Section 3.1 dedicated to the third question, and Section 3.2 concerns the second question. Sections 3.3 and 3.4 treat the third question, and Section 3.5 concerns the fourth question. The main results are presented in five publications (listed above). These results are referred to briefly in the text, so the reader should turn to the original publications listed in the appendices for this part of the results. Chapter 4 aims at enlarging the discussion about the equalization and prefermentation and gives the Finnish point of view of the feasibility of these pre-treatment processes. The reliability and validity of the results are also discussed and recommendations for further research are given. Finally, the main contribution from the study is summarized in Chapter 5 - Conclusions.

2 Materials and methods

2.1 Introduction

This section describes the process and operational procedure at the Pihlajaniemi WWTP and the sewer network connected to the plant. The set-up for the research project and the data collection campaigns are also described. In this study, the volume of the existing primary clarification was used for two purposes – firstly, to level out flow variations in the biological reactors and secondly, to produce volatile fatty acids (VFA) needed in the BNR process. A short trial for prefermentation in the primary clarifier basins had already been carried out at the plant at the end of 1990s. This trial had left the plant operators quite sceptical because of accumulation of sludge in the primary clarifiers, odour and H_2S toxicity (Tahvanainen, 2002). For this reason, prefermentation was implemented without fully ceasing from the raw sludge removal, in spite of the loss in prefermenter efficiency. Moreover, the plant did not have any additional basin volume available, which could have allowed off-stream prefermentation. Thus, prefermentation was performed in the clarification basins simultaneously with the flow equalization. In other words, flow equalization was the main focus, and the possibility of low-yield prefermentation in the same basin was investigated.

2.2 Full-scale wastewater treatment process

The Pihlajaniemi WWTP in the City of Savonlinna was constructed in 1978 as a conventional low-loaded activated sludge plant with simultaneous precipitation of phosphorus. The original plant was designed for a population equivalent (PE) of 33,000. The original process consisted of screening, sand and grease removal, primary clarification, aeration, secondary clarification and chlorination. It was dimensioned for a mixed-liquor suspended solids (MLSS) of 3.5 kg/m^3. Due to the high load from pulp manufacturing, the plant was not always in compliance with the effluent standards of the time, and foaming occurred regularly. In the beginning of the 1980s, a tertiary treatment with dissolved air flotation (DAF) and sand filtration was added in order to improve the treatment performance. Pulp manufacturing in Savonlinna closed down in 1985, but the tertiary treatment was kept in operation. Nowadays, DAF is in operation only when turbidity in the secondary clarifier effluent is above 10 FTU. Otherwise, tertiary treatment acts as conventional sand filtration.

Nitrification was introduced to the activated sludge process of the plant in 1987 by increasing the MLSS concentrations to 6–10 kg/m^3 in the aeration tanks. In 1991–1993 the plant was modified for nitrogen removal, and in 1994–1995 it was modified for both biological phosphorus and nitrogen removal. The process configuration developed was called the HUT-Savonlinna-process (Kiuru and Rautiainen, 1998). Nowadays, the nutrient removal process is mainly operated with a configuration of four zones: anaerobic, anoxic, aerobic and anoxic (ADND) zones, although AND and ADN configurations are also possible. The last anoxic zone was later added to the process in order to minimize the oxygen concentration in the recycle flow to the first anoxic zone. The volumes of the anaerobic zone and anoxic zone can each be adjusted between one-eight and one-quarter of the biological reactor volume. Thus, the aerated volume, where nitrification takes place, varies from three-eighths to five-eighths of the reactor volume. Usually the larger aeration volume is used during cold water periods, from about October to May. Since the end of the 1990s, the plant has been operated at MLSS concentrations varying between 3 and 8 kg/m^3. The sludge concentration was lowered due to bulking sludge, which was assumed to be the consequence of high sludge concentration. Biological phosphorus removal is completed with chemical precipitation. Small dosage of ferrous sulphate is added at the beginning of secondary clarification. More detailed information about the plant can be found in appendices I and V. The sewer network of the City of Savonlinna is described in Appendix V.

Information about the plant influent flow and load can be found in I, II and V. During the study, the plant was not heavily loaded – total aeration basin volume is 3000 m^3, giving a volumetric loading of 0.2-0.6 kgBOD/m^3/d and sludge loading of 0.05 - 0.1 kgBOD/kgMLSS/d. The surface loading in the secondary clarifier varied between 0.2 and 0.4 m/h. The plant does not have any numerical effluent requirements for nitrogen, but the permit requires removal of ammonium as efficiently as possible. Due to the effluent standards, the plant is operated with phosphorus removal as the highest priority.

2.3 Earlier measures to improve the feed water at the plant

The influent flow rate at the Pihlajaniemi WWTP varied relatively little already before the study because efforts to level out the variation of the influent flow and load were made several years before this research project. Special attention has been paid to control the recycle flows inside the plant. The reject and wash waters from different processes at the plant are led either to the plant inlet or to the sand removal basin. Backwash water from flotation filters (representing almost 10% of the dry weather plant influent) evens out the diurnal flow variation, since the filter wash cycle always takes place during the night. Reject water from sludge thickeners and centrifuge account for a smaller portion than backwash water, being 5% and 1% of the dry weather plant influent respectively. They arrive at the plant inlet whenever their storage well is full. In addition, the wide-spread sewer system connected to the Pihlajaniemi WWTP flattens out the flow variations. The sewer system has been expanded significantly during the previous 20 years, as described in Appendix V. At the time of the study, only the sewage from the neighbouring town of Punkaharju was pumped to the plant via a storage tank. No significant storage volume existed elsewhere in the system, e.g. in pumping stations.

The lack of suitable organic matter has also been observed several years before the study, and the partial primary clarification bypass was in operation in order to increase the organic load in the feed water of the BNR process. The reject water from sludge thickeners and centrifuge contained high concentration of soluble organic matter, but the reject flows were not time-controlled.

2.4 Modifications in the primary clarification during the study

Equalization and prefermentation process train

During the 18 months of the study, one pair of primary clarifiers was modified in the equalization basin. During the first phase of the study (January–August, 2003), only the equalization pump was in operation. During the second phase of the study (September 2003 to March 2004), a recycle pump was added to the basin in order to enhance prefermentation (see Figure 2.1). The raw sludge removal was kept in operation, but it was decreased by 50% during the second phase. The set-up and the operation of the equalization/prefermentation basin are described in appendices I and III. In the text, the process train where this pre-treatment was used will be referred to as the equalization/prefermentation process train.

Primary clarification process train

The other pair of primary clarifiers was operated as earlier at the plant, i.e., 15-20% of the influent flow bypassed the primary clarification. This allowed the comparison of the results from the research period with the collected operator data from previous years. The objective of the bypass was to increase the amount of organic carbon fed to the BNR process, but it provided some equalization of flow as well. Throughout the text, the process train with this pre-treatment will be referred to as the primary clarification process train.

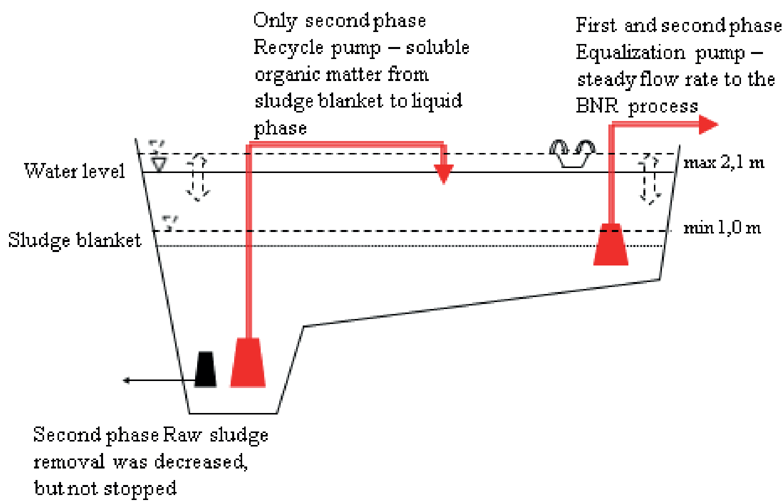

Figure 2.1 Operation principle of equalization pump and recycle pump.

2.5 Control of the equalization/prefermentation basin

The submerged equalization pump was controlled by two water level measurements: one in the tank and the other in the overflow channel. The pump was stopped at the minimum water level (1.0 m) in order to keep the submergence needed for the cooling of the pump. When overflow to the channel occurred (2.1 m), the pump was stopped in order to avoid unnecessary pumping and restarted again when the flow in the overflow channel decreased. With regular overflow from the basin, no accumulation of scum occurred. The pump was operated at a steady speed. The submergence depth of the pump changed with the fluctuating water level in the basin, which in turn affected the rate of the pump. The rate varied between 27 l/s (minimum water level) and 48 l/s (maximum water level). The fact that the flow rate of the pump increased as the water level in the basin increased provided an additional control smoothing the flow rate to the biological reactor. The water level measurement data in the equalization/prefermentation basin was collected in order to analyze the operation of the flow equalization. The operation time of the equalization pump was registered as well. The recycle pump in the second phase of the study was operated at a steady speed.

2.6 Operational parameters during the study

During the study, the plant was operated as if there were two different treatment processes: one with equalized VFA-rich feed and the other with feed consisting of 20% raw wastewater and 80% conventionally pre-clarified wastewater. In order to compare the performance between the two process trains, the process parameters were intended to be kept at the same level for both trains. Table 2.1 shows the different controllable process parameters and the objective of the control.

Table 2.1 Controllable process parameters and objective of control.

Process parameter	Objective of control
Mixed liquor suspended solids (MLSS)	MLSS 5–8 kg/m^3 during the cold period in order to increase the biomass in the process. MLSS 3 – 5 kg/m^3/d in summertime To be kept same for both process trains during the study.
Return activated sludge (RAS)	NO_3 concentration in RAS 1–2 mg/l Could not be set for each process train separately.
Nitrate recycle	Could not be set for each train separately.
Waste activated sludge (WAS)	Sludge age (>12d) Sludge age to be kept the same for both trains during the study.
Size of the aerated zone	Aerated zone increased during the cold period in order to achieve maximum nitrification. Same for both trains during the study.
Dissolved oxygen (DO) concentration in the aeration basin	2.3-2.5 mg/l in the beginning of the aerated zone, 2.3 mg/l towards the end of the zone Same oxygen concentration for both trains during the study.

The sludge age (SRT) and MLSS during the study is presented in Figure 2.2. The MLSS concentration varied between the process trains, the average concentrations being 4.9 kg/m^3 in the equalization/prefermentation process train and 4.8 kg/m^3 in the primary clarification. The SRT was approximately similar between the trains, except during July-August and October-November, when the sludge age increased in the equalization/prefermentation train. During this time, the SRT in the equalization/prefermentation process train was above 20 days, whereas in the primary clarification process train it was around 10 days. On average during the whole study, the sludge age in both process trains were similar: 14 days in the equalization/prefermantation process train and 11 days in the primary clarification process train. Due to poorer settleability of sludge in the primary clarification process train, more sludge escaped in the effluent, which in turn decreased the SRT. Moreover, especially in August and September, only a small amount of sludge was wasted from the equalization/prefermantation process train because the sludge yield was very small. The sludge age was calculated with the following equation.

$$\text{Sludge age (SRT)} = \frac{V_{aeration} * X_{aeration}}{Q_{WAS}*X_{WAS} + Q_{effluent}* X_{effluent}} \qquad (1)$$

where $V_{aeration}$ is the volume of the aeration basin (m^3)
$X_{aeraton}$ is the sludge concentration in the aeration basin (kg/m^3)
Q_{WAS} is the flow rate of the waste activated sludge (m^3/d)
X_{WAS} is the sludge concentration of the WAS (= $X_{aeration}$ in this case)
$Q_{effluent}$ is the effluent flow rate (m^3/d)
$X_{effluent}$ is the concentration of suspended solids in the effluent (kg/m^3)

The sludge age referred to in Appendices II and IV is calculated taking into account also the sludge in the secondary clarifiers.

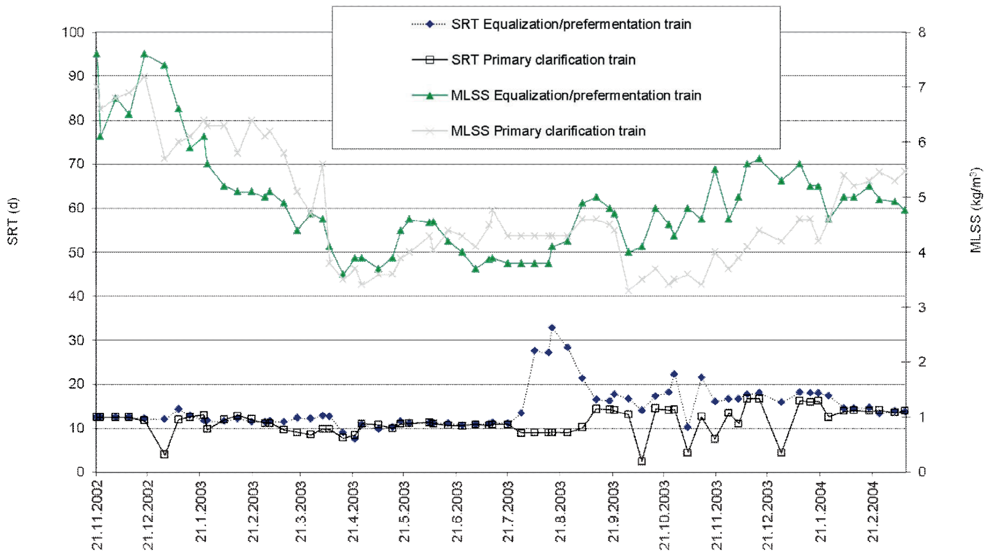

Figure 2.2 SRT and MLSS in both process trains during the study.

The waste-activated sludge (WAS) was removed from the aeration basin. The RAS flow rate and nitrate recycle flow rate could not be controlled separately for each process train, and their control was a compromise between the two process trains. As planned, the size of the zones and DO set-points were similar for both process trains. During spring and summer 2003, the DO set-point was 2.5 mg/l in the beginning and 2.3 mg/l in the second zone, but from autumn 2003 onward, 2.3 mg/l was used everywhere.

2.7 Sampling campaign

The plant was operated with the modified pre-treatment during 24 months. The process was modified eight weeks before the sampling was started in November 2002 in order to allow for some acclimation. The analytical part of the study was confined to 18-month monitoring of the influent wastewater and the pre-treated water, activated sludge reactors and secondary clarifier effluent from two differently operated process trains. The study schedule is presented in Table 2.2.

Table 2.2 Study schedule

	Operation	Time period	Sampling
Phase I	Equalization, cold weather	January–March 2003	Extended weekly monitoring + 6- to 8-week sampling campaigns
	Equalization, warm weather	June–August 2003	
Phase II	Equalization + enhanced VFA production, warm weather	August–November 2003	
	Equalization + enhanced VFA production, cold weather	December 2003–February 2004	

The plant had a process monitoring program consisting of a weekly time proportional 24h-composite sampling from plant influent, secondary clarifier effluent and effluent wastewater after flotation filters and grab sampling from different zones in the biological reactor and return sludge. The sampling points are shown on the process scheme in Appendix V. The weekly sampling was done on a rotating weekday (Monday–Thursday), and every fifth week a three-day composite sample (from Friday morning to Monday morning) was taken in order to measure the weekend conditions. During the study the monitoring program was expanded with composite and grab samples taken separately from both process trains. In addition, samples from the primary clarifier effluent were taken from both process trains.

In addition to the extended weekly process monitoring, four sampling campaigns were carried out during the study, two during each phase in different conditions. During these campaigns, composite samples were taken three times a week, of which one was during the weekend. The objective was to monitor the diurnal variations of the process. Time proportional 3h-composite samplers were used over 24 hours. Due to the limited number of samplers, all sampling points were not collected every time as 3h-composite samples, but the rest of the samples were 24h-composite samples. Sampling campaigns included influent wastewater and pre-treated wastewater samples as well as secondary effluent samples from both process trains. Moreover, grab samples were taken from the influent and pre-treated wastewater for readily biodegradable COD (RBCOD) measurement as well as different zones in the biological reactor for PO_4-P measurements. Each campaign lasted 6 to 8 weeks. The sampling campaigns were intended to be scheduled during dry weather periods, although rainy days could not be completely avoided in summer and autumn.

In September 2004, an additional sampling campaign was carried out in order to study possible secondary phosphorus release from the secondary clarifiers. Grab samples were taken from the sludge blanket at the beginning, in the middle and from the end of the rectangular clarifier basin at different depths. Altogether 24 samples were taken during one day between 9 a.m. and 1 p.m.

The sampling points and related analyses are listed in the following:

- **The influent wastewater samples** were taken from the beginning of the sand removal basin after screening. The 24h-composite samples were collected to a refrigerated sampler. The parameters analyzed are listed in Appendix V. The influent flow rate was measured on-line.

- **The pre-treated wastewater samples** were taken from the channel between the pre-treatment and biological reactor. The grab samples for RBCOD measurement were taken around 9 a.m. The parameters analyzed were the same as from the plant influent (V). The flow rate in each process train was measured by Venturi flow meters in the channels between the pre-treatment and biological reactor. The wastewater for each process train was distributed by over-flow weirs in the sand removal basin. In spite of recurrent adjustment of the weirs, the flow distribution between the process trains was not equal – 54% of the total flow was led to the equalization/prefermentation process train and 46% to the primary clarification process train.

- The wastewater samples were taken from **different zones in the biological reactor**. The parameters analyzed in the weekly process monitoring program and during the sampling campaigns are listed in Table 2.3.

Table 2.3 Samples from different zones in the biological reactor.

Weekly monitoring program	Grab sample	SVI, DSVI, MLSS, NO_3-N in RAS
Sampling campaigns	Grab sample	PO4-P from each zone in the biological reactor

Several on-line measurements were used to monitor the biological reactor. Sludge concentration was measured on-line from both process trains. Each process train had two measurements for oxygen concentration, which were used to adjust the air flow from the blowers. Air flow was measured from each aerated zone. An on-line analyzer for NH_4-N was used in the end of the activated sludge basin measuring samples from the process trains. RAS and WAS flow rates were measured, and on-line NO_3-N analyzer was used to measure nitrate concentration in the RAS. The analyzer was only measuring from the primary clarification process train. From the other process train, grab samples were taken.

- **The secondary clarifier effluent samples** were taken separately from both process trains and from the channel before the flotation filters, where wastewater from both process trains was mixed. The parameters analyzed in the weekly process monitoring program and during the sampling campaigns are listed in Table 2.4.

Table 2.4 Samples from secondary clarifier effluent.

Weekly monitoring program	24-hour composite sample	BOD_7(ATU), COD_{Cr}, P_{tot}, N_{tot}, NH_4-N, PO_4-P, NO_3-N, SS
Sampling campaigns	3/24-hour composite sample	COD_{Cr}, NH_4-N, NO_3-N, PO_4-P, SS

On-line PO_4-P and turbidity measurements were done from the secondary clarifier effluent. The turbidity analyzer controlled flotation air addition and the PO_4-P measurement controlled the polymer addition before the flotation filters. The sludge blanket in the secondary clarifiers was measured manually twice a day from each clarifier basin. During one day, NO_3-N, PO_4-P and oxidation-reduction potential (ORP) profiles were also measured with grab samples from the sludge blanket in the secondary clarifiers.

The analyses were mainly performed following Finnish standard methods. All the methods used and analyzers at the plant are listed in Table 2.5. The soluble COD_{Cr} and BOD_7(ATU) were determined with the standard method from filtered (GF/A) samples. The soluble COD production rate was determined with the difference of soluble COD concentration in the primary basin influent and effluent divided by the hydraulic retention time. The sludge in the aeration basin was analysed with a microscope approximately one a month.

Table 2.5 Methods and analyzers.

Analyses	Standard/method
Total phosphorus	SFS 3026/86
Total nitrogen	SFS 3031+Lachat
COD_{Cr}	SFS 5504
Suspended solids	SFS-EN 872
PO_4-P, filtered GF/A	SFS 3025/86 (Savolab) Hach DR/2000 (Pihlajaniemi)
NH_4-N	SFS 3032+Lachat (Savolab) Hach DR/2000 (Pihlajaniemi)
NO_3-N	SFS3030+Lachat (Savolab) Hach DR/2000 (Pihlajaniemi)
BOD_7 (ATU)	SFS 5508/91
VFA (the sum of acetate, propionate, butyrate and isobutyrate)	Gascromatography HP FFAP 19095N-123, model 7590A, FID (Sørensen et al., 1991)
RBCOD	UCT (Wentzel et al., 1995)

3 Equalization and prefermentation from tap to plant effluent

3.1 What happens in the pre-treatment?

3.1.1 Introduction

In this chapter, first the pre-treatment units (primary clarification, prefermentation and equalization) are described in detail. Screening and sand and grease removal are not included. Secondly, the feed water characteristics that are considered beneficial for BNR are introduced. In the end, the results of the performance of the pre-treatment at the Pihlajaniemi WWTP are presented. Pre-treatment units may have several different and contradictory goals, even if the overall objective is to improve the performance of the biological treatment. Also, the goals may have changed over time. For example, the traditional objective of primary clarification is to remove as much of the organic load as possible, but the requirements for sufficient carbon feed to achieve BNR have made the trade-off more difficult. Similarly, a separate fermentation unit can produce readily biodegradable organic matter and VFAs for the needs of BNR process, but it usually increases the overall organic load to the process.

3.1.2 Primary clarification

3.1.2.1 Objectives and efficiency of primary clarification

The objective in the primary clarification is to remove the excess load of organic matter before the biological process as primary sludge. By reducing the load on the biological part, energy savings are achieved by decreased aeration air requirements. Also, the necessary volume in the nitrification step can be reduced (Æsøy and Ødegaard, 1994). Furthermore, excess sludge yield from the biological process can be decreased. The excess sludge is usually voluminous and more difficult to dewater (Rössle and Pretorius, 2001). Moreover, primary clarification can be advantageous in the removal of inhibiting constituents for the biological treatment.

The efficiency of the primary clarification can be expressed by the removal of suspended solids and organic matter. In the primary clarifier, BOD and COD removal are usually considered to be about 35%, solids are reduced by about 60%, and nitrogen and phosphorus by about 10% (Harremoës et al., 1993). The removal of particulate substrates depends on residence time and will not be further improved after more than three hours (Otterpohl and Freund, 1992). In low temperatures, water viscosity increases and particle settling is retarded (Metcalf and Eddy, 1991, p. 474). Moreover, the quality of the particulate matter influences the settling characteristics.

3.1.2.2 Primary clarification from the point of view of the BNR process

Primary clarifiers have traditionally been considered by taking into account only sedimentation. Typically, the type of settling in primary clarifiers is flocculent particle settling, which is characterized by the flocculation of solid particles as they settle through the water column (Takács et al., 1991). In the Takàcs clarifier model, the settler is divided into a number of layers and a mass balance is made over each layer to evaluate the SS profile in the settler. Nevertheless, several other phenomena are taking place in primary clarifiers in addition to sedimentation of particulate organic matter. Gemaye et al. (2001) observed that ammonification, hydrolysis and flocculation can occur in primary clarifiers. These processes may take place especially in clarifiers with a high hydraulic retention time or when anaerobic digester supernatant or excess biological sludge is recycled to the primary clarifier. Ribes et al. (2002) proposed a model that includes sedimentation, compression and biological processes. Furthermore, during wet weather flows, even peaks of COD and nitrogen load that are higher than the influent can occur in the primary clarifier effluent. This is because the content of the primary clarification basin is transported to a biological basin in a short time (Harremoës et al., 1993; Niemann and Orth, 2001).

From the point of view of biological nutrient removal, primary clarification can have a detrimental effect on the desired wastewater characteristics, especially if only sedimentation is considered. Indisputably, the nutrient to carbon ratios increase due to a larger COD settling and removal rate compared to nitrogen and phosphorus (Harremoës et al., 1993), but the quality of the organic matter is completely overlooked. Therefore, it seems that biological processes (e.g. hydrolysis in the primary clarifiers) should be considered, but studies quantifying the VFA production in primary clarification basins could not be found.

3.1.3 Prefermentation

3.1.3.1 Objective and definition

In municipal wastewater, the available organic matter is frequently a limiting factor for BNR. Often, the missing organic matter is added to the process as external carbon, e.g. methanol, ethanol, acetate, glucose or sugar (Tam et al., 1992; Canziani et al., 1995; Swinarski et al., 2009). Nevertheless, the total amount of carbon in raw wastewater is usually sufficient to fulfil the needs of BNR processes. The problem is that most of the carbon is in slowly biodegradable form and thus not directly available for denitrifying bacteria nor PAOs. In wastewater treatment hydrolysis refers to the breakdown of complex organic compounds by means of extracellular polymers and the conversion into smaller products that can subsequently be taken up and degraded by microorganisms (Morgenroth et al. 2002). Because identification and description of these reactions in wastewater is impossible, they are conveniently collected into an overall single hydrolysis mechanism (Insel et al. 2002) By applying hydrolysis, it is possible to convert slowly biodegradable organic matter into readily biodegradable form. . The biological hydrolysis consists of four phases:

1) The volatile suspended solids are hydrolysed by means of extracellular enzymes.
2) In the acidogenic phase, the products from the hydrolysis are fermented into volatile acids.
3) In the acetogenic phase, high molecular fatty acids as well as volatile acids are decomposed.
4) Methane is produced in the methanogenic phase (Brinch et al., 1994).

The sequence of the processes occurring during hydrolysis is shown in Figure 3.1.

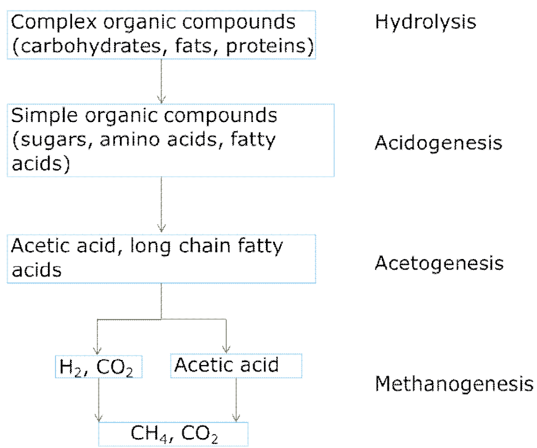

Figure 3.1. Diagram of the processes in biological hydrolysis.

The conversion of complex readily biodegradable substrate to VFAs under anaerobic conditions is commonly termed "fermentation". In wastewater treatment this technique has been used for the past 20 to 30 years, and it is usually called prefermentation. In the process, the two first steps of biological hydrolysis are taking place. As the fermentation products are soluble, they can be separated from the fermented sludge as VFA-enriched supernatant (Canziani et al., 1995). Münch and Koch (1999) have determined prefermentation as:

> "the intentional anaerobic production of VFA in primary treatment tank(s), from suspended or settled organic matter present in domestic and industrial wastewater, with the aim of transferring and using these VFA to improve the biological nutrient removal performance of a WWTP."

In the case of enhanced biological phosphorus removal, prefermentation is aiming specifically at producing VFAs, although they can be formed in the anaerobic zone from complex RBCOD, whereas for denitrification, the goal is to increase the fraction of the readily biodegradable organic matter in general.

3.1.3.2 Prefermenter configurations

Most often, the prefermentation is applied to primary sludge or raw wastewater (Banister and Pretorius, 1998; Münch and Koch, 1999). Applications in which a prefermenter receives the entire wastewater flow are called in-line prefermenters (Rössle and Pretorius, 2001). They are often making use of primary clarifier basins and are also referred to as activated primary treatment (APT) (Münch and Koch, 1999). Usually the primary sludge is recycled to the APT tank influent in order to increase the SRT and create anaerobic conditions (Wedi, 1992; Rössle and Pretorius, 2001), but applications with primary sludge recycled to the end of the basin have also been reported (Christensson et al., 1998; Jönsson et al., 1996). Cuevas-Rodrigues et al. (1998) presented an application with sequencing batch reactors. The term 'side-stream prefermenter' is usually used when only primary sludge is fermented. Prefermentation has also been studied with RAS (Jönsson and Jansen, 2006; Vollertsen et al., 2006), with chemical sludge (Kristensen et al., 1992; Æsøy and Ødegaard, 1994) and with WAS (Yuan et al., 2009) or a mixture of primary sludge and WAS (Moser-Engeler et al., 1996). Hydrolysis can be performed biologically, chemically and physically (thermal hydrolysis), or by various combinations of these methods, although the biological method seems to be prevailing (Skalsky and Daigger, 1995).

3.1.3.3 Measuring the efficiency and parameters affecting the prefermentation

The efficiency of prefermentation is commonly measured with the "rate of VFA production" r_{VFA}, which for a continuous prefermenter at steady state is calculated as follows:

$$r_{VFA} = \frac{C_{VFA}^{eff} - C_{VFA}^{inf}}{HRT} \quad (2)$$

Where r_{VFA} Rate of VFA production (mg/l/h)

$C_{VFA}^{eff/inf}$ VFA concentration in prefermenter influent/effluent (mg/l)

HRT Hydraulic retention time of the perfermenter (h)

In order to compare different prefermenter configurations, a parameter describing the VFA concentration increase in the total wastewater flow, ΔC_{VFA}, has been proposed. It is calculated as follows:

$$\Delta C_{VFA} = f_{QP/QT} \cdot C_{VFA}^{eff} \quad (3)$$

Where ΔC_{VFA} VFA concentration increase in total wastewater flow due to prefermenter (mg/l)

$f_{QP/QT}$ Fraction of total plant flow that is being sent to prefermenter (= 1 for in-line prefermenters) (Münch and Koch, 1999).

This equation is valid for prefermenters fed with raw wastewater and with the assumption that the VFA concentration in the raw wastewater are close to zero.

In the case of denitrification, the performance of the prefermenter can be measured with the degree of solubilisation expressed as the ratio of soluble COD produced to influent VSS or as the ratio of soluble COD produced to influent particulate COD (Barajas et al., 2002). Full-scale observations have shown that with prefermentation of primary sludge, VFA yields of approximately 10% of the total influent COD concentration can be expected with sludge retention times of less than 6 days (Banister and Pretorius, 1998). Münch and Greenfield (1998) introduced a relationship between the VFA concentration and pH value in prefermenter that the VFA concentration to be estimated from a single point pH measurement.

The VFA production is affected by HRT, temperature, pH, sludge concentration, mixing of sludge, and the amount of sludge recycle. The VFA production increases with increasing HRT until methanogenesis starts and decreases the VFA production. The optimum HRT value depends on the particular type of the wastewater. (Banerjee et al. 1998) In general, VFA production increases with increasing temperature, although examples of the contrary also exist. The pH values around neutral usually give the highest VFA concentrations (Wedi, 1992; Christensson et al., 1998; Teichgräber, 2000; Chanona et al., 2006). Barajas et al. (2002) also reported that covering the APTs allowed for better control of temperature and ORP and resulted in improved fermentation. Ruel et al. (2002) explained the influence of temperature by the fact that lipid and protein solubility becomes higher as the temperature rises. Moreover, some fermentative bacteria cannot grow at low temperatures (below 14 °C), leaving some substrates not fermented. On the other hand, Banerjee et al. (1998) showed that with some sludges, an increase in temperature (to above 25 °C) can result in poor settling characteristics of the sludge and thus a poor performance of the perfermenter. Moreover, Barnard and Steichen (2007) stated that prefermenters in cold areas perform better than those in warmer areas, producing up to 0.3 g VFA/g VSS applied as opposed to values such as 0.1 g/gVSS in warmer areas. This could be attributed to less fermentation taking place in the sewer system, leaving more readily biodegradable matter in the primary sludge. VFA yields were found to increase in diluted primary sludge (less than 2%), and mixing proved to decrease the prefermentation performance (Banister and Pretorius, 1998). Furthermore, seeding the prefermenter with partially fermented sludge has been proven to boost VFA yields. The effect of sludge elutriation rate (SER) and sludge recycle rate (SRR) in the case of in-line prefermenters have not been studied very intensively. The SER refers to the ratio of the mass of fermented solids recycled to the volumetric inflow rate of the raw wastewater to the APT, whereas the SRR refers to the volume of sludge recycled to the volumetric inflow rate of the raw wastewater to the APT (Rössle and Pretorius, 2001). Recently, Bouzas et al. (2007) showed that higher SRR increased the VFA production. Chanona et al. (2006) introduced an optimization algorithm for controlling the effluent VFA concentration by the raw sludge wastage and SRR.

3.1.3.4 Side-stream vs. APT performance

According to Münch and Koch (1999), the side-stream prefermenters in general have a higher rate of VFA production than the in-line prefermenters, 4–28 mg/l/h and 1.4 mg/l/h respectively. Rössle and Pretorius (2001) suggested higher values (15–70 mg/l/h for side-stream and less than 10 mg/l/h for APT tank), but the authors did not give any references. For APT application, solubilisation of 66 mg soluble COD/g influent particulate COD has been reported (Barajas et al., 2002). In spite of the lower VFA production rate, in-line prefermenters offer a competitive option because the entire wastewater flow is treated and a small increase in VFA concentration can represent a large mass of VFAs produced (Münch and Koch, 1999). Nevertheless, Bajaras et al. (2002) observed that in an in-line fermentation basin, only 31% of the influent VFA potential was reached. The authors stated that VFA production was moderate because most of the influent readily fermentable COD and a fraction of solids were not in contact for long periods of time with fermenting sludge. On the other hand, the acidogenic phase, during which the VFAs are produced, can also take place in the anaerobic zone of the EBPR process (Christensson et al., 1998), and when this is the case, the accomplishment of the fermentation process until acidogenesis is not so meaningful. Wedi (1992) reported on an unsuccessful full-scale in-line prefermenter, where no VFA production was observed. Moreover, accumulated primary sludge was high in viscosity and caused breakdowns of elutriation pumps. Also, when hydraulic loading increased, a large mass of sludge was washed out and caused shock loading of the activated sludge process. Also, Pitman (1999) reported on excessive solids carry-over and recommended the use of separate fermentation and elutriation tanks. On the other hand, Teichgräber (2000) pointed out the risk of odour development and the risk of development of poisonous or explosive gases, based on German experiences with side-stream prefermenters. Vollertsen et al. (2006) pointed out the possibility of using the side-stream hydrolysis tank for equalization of (for example) nitrogen-rich reject water from the dewatering of sludge.

3.1.3.5 VFA speciation

VFA-COD equivalent indicates the proportion of longer-chain VFAs, which is usually between 1.067 (equivalent of acetic acid) and 1.514 (equivalent of propionic acid). It seems that the VFA speciation is also affected by the configuration of the prefermenter. Münch and Koch (1999) reported a clearly lower (1.15) VFA-COD equivalent for APT prefermenter than side-stream prefermenters (1.24–1.40). Rössle and Pretorius (2001) collected typical VFA composition distributions from prefermenter effluents from several countries. Acetic acid was produced predominantly with the fraction varying between 38% and 71%, followed by propionic acid between 24% and 45%, and butyric acid between 0% and 16%. Valeric acid was quite rare (0%–10%). The VFA-COD equivalent varied between 1.22 and 1.38 g COD/g VFA. Unfortunately, the type of prefermenter was not reported and all the references were not accessible.

Æsøy and Ødegaard (1994) observed that the fraction of acetic acid was decreasing and the fraction of butyric acid increasing with increasing sludge volatile solids concentration. The authors stated this to be the result of the increasing concentration of hydrogen gas, which influences the energetic conditions. The authors also reported that 84% of the VFAs originated from protein materials. Gonçalves et al. (1994) reported that in their experiments, 90% of the VFA was in form of acetic acid. They also pointed out that a pH close to neutral encouraged the transformation of acetic acid to acetate, which is non-volatile. Bouzas et al. (2007) confirmed the earlier findings of the influence of sludge concentration on the type of VFA produced. They also stated that the conversion of longer chain VFA into acetic acid can only take place at a low hydrogen partial pressure.

3.1.3.6 *Nutrient solubilisation*

Nitrogen and phosphorus are released during hydrolysis. Table 3.1 shows nutrient solubilisation values reported in the literature. Banister et al. (1998) observed that nitrogen and phosphorus release depended on the retention time in the fermentation, solids concentration and wastewater composition. Bajaras et al. (2002) and Christensson et al. (1998) reported a small decrease in the fermenter effluent pH as a result of VFA production. The change in pH will be strongly affected by the level of alkalinity, i.e., the buffering capacity in the influent water. A small increase in alkalinity was also observed, which was attributed to ammonification. Christensson et al. (1998) observed that the phosphorus solubilisation in the fermenter was accompanied by a corresponding release of metal ions. The authors concluded that the released phosphorus could be expected to be precipitated again under oxidizing conditions. This assumption was also verified in a laboratory batch experiment. The authors stated also that even though the iron and calcium concentration in the influent wastewater was low, iron or calcium can be supplied to the fermenter influent from other parts of the process.

Table 3.1 Nutrient solubilisation values reported in the literature.

NH_4-N solubilisation	PO_4-P solubilisation	Reference
0.09 mg NH_4-N/mg acetic acid	0.035 mg PO_4-P/mg acetic acid	Banister and Pretorius (1998)
0.05–0.14 mg NH_4-N/mg VFA	0.06-0.09 mg PO_4-P /mg VFA	Skalsky and Daigger (1995)
0.02–0.06 mg NH_4-N/mg COD hydrolysed	0.005 mg PO_4-P /mg COD hydrolysed	Moser-Engeler et al. (1998)
0.002 – 0.007 g NH_4-N/g influent COD	0.001–0.005 g PO_4-P/g influent COD	Barajas et al. (2002), in-line fermenter

3.1.4 Flow and load equalization

3.1.4.1 Objective and benefits

The primary objective of the flow equalization basins has been to dampen the variations in the flow to achieve nearly constant flow rates through the downstream treatment processes. Moreover, equalizing the organic load to the process could significantly benefit the performance of the biological treatment process (Dold et al., 1984; Armiger et al., 1993). There is an interest in levelling out the variations in the feed water, because changes in flow rate already in the scale of normal diurnal variation can cause problems with aeration control and other control systems at the plant. On the other hand, during low flow rate, the micro-organisms in the reactor will suffer from starvation. Van Loosdrecht et al. (1997) pointed out that these feast-famine conditions would lead to development of a bacterial population capable of using storage polymers, but the growth rate on storage polymers would be clearly lower than on the original substrate. Harremoës et al. (1993) pointed out that diurnal variations of flow and pollutant loads are often larger in small plants because of shorter sewer systems. Furthermore, the diurnal load variations of BOD and phosphorus are often larger than the variations of the nitrogen load, meaning that the C/N and C/P ratios, which may be considered important for biological nutrient removal processes, change throughout the day. This is because different pollutants (e.g. organic matter, ammonium, and phosphates) are generated from different sources within a household. The hour of usage of each appliance and their proportion of the whole wastewater flow at the time determine the concentration pattern in the wastewater discharge (Friedler and Butler, 1996).

With a larger scale of variation during wet weather flows, the effects to the process will be magnified: the activated sludge will be transported into the secondary clarifier leading to a reduction in purification capacity (Jack and Ashley, 2002). Also, the aeration capacity may be exceeded, and consequently, a reduction in nitrification capacity may occur. Moreover, solid content in the secondary clarifier effluent may increase as a result of hydraulic overloading (Dold et al., 1984; Pons and Corriou, 2002). Furthermore, higher concentrations of particulate substances in the final clarifier effluent will affect the sludge age (Otterpohl and Freund, 1992).

Further benefits of equalization basins are the dilution of inhibiting substances (Metcalf and Eddy, 1991, p. 205) and, in the case of storm water tanks, the dampening of temperature shocks. Canziani et al. (1995) pointed out that influent load fluctuations can also impede the prefermentation and necessitate internally produced RBCOD to be supplemented by external substrate. Dold et al. (1984) pointed out that substantial benefits from both load and flow equalization are expected only with long sludge age. Also, according to the authors, effluent COD and BOD are insensitive to influent load variations because the particulate material is enmeshed and adsorbed by the sludge mass. Thus, the performance of the process should not be measured as effluent COD content.

3.1.4.2 Different configurations to reach flow equalization

Flow equalization can aim at levelling out the diurnal variations or the wet weather flow peaks. Wet weather conditions require a different operation of the equalization basin and thus diurnal flow control and storm water control should be dealt with separately. Usually the storm water basin, referred to as a storage tank, is filled only when the maximum flow capacity of the biological reactor is exceeded after the rain event and then emptied progressively, whereas an equalization basin aiming at diurnal flow control is filled and emptied every day. Pons and Corriou (2002) observed that the effluent quality was clearly improved more with an equalization basin than with a storage tank. This observation was supported by Funamizu et al. (2000) and Jack and Ashley (2002), although the authors stress the fact that high shock loads, which can be avoided with the use of a storage tank, can be more harmful for the receiving watercourse than fairly dilute pollution. Nevertheless, the cost of pumping was also higher with the equalization basin (Pons and Corriou, 2002). On the other hand, Dold et al. (1984) suggested that pumping costs could be decreased, without reducing the equalization efficiency, by up to 60% of the daily inflow bypassing the equalization basin. Nielsen et al. (1996) proposed using an aeration basin as the clarification tank during short periods in order to improve the hydraulic control of secondary clarification during wet weather flows.

Diurnal flow equalization can be arranged in-line or off-line. In the in-line configuration, all of the flow passes through the equalization basin. In the off-line arrangement (also called a side-stream basin), only the flow rate above some predetermined flow rate is diverted into the equalization basin. With an off-line arrangement, the pumping requirements are minimized, but the amount of pollutant concentration damping is reduced compared to an in-line configuration (Charlton, 1994; Metcalf and Eddy, 1991, p. 204). Recently, Murakami et al. (2008) proposed a configuration for equalization with the principle of dividing the influent flow into three flows, which are each divided into two equal parts and then joined again into three parts (Figure 3.2).

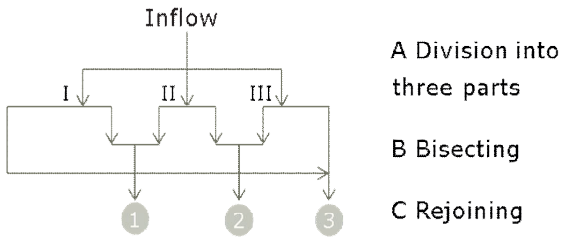

Figure 3.2 The principle of the equalizing distribution tank (Murakami et al. 2008)

3.1.4.3 *Successful design and operation of equalization basin*

Although experienced plant operators can achieve satisfactory results in manually controlling the outflow from an equalization basin, manual control is labour-intensive and prone to poor repeatability (Mather and Shaw, 1993). Several successful control strategies have been reported (Dold et al., 1984; Mather, 1997). Dold et al. (1984) made calculations based on influent data from several plants with an equalization algorithm and concluded that the optimal size for an equalization basin is a volume allowing for a 4- to 6-hour mean retention time based on the mean inflow rate. They also introduced a control strategy that predicted the expected influent patterns based on historical data. Then, the outflow profile was computed for the ensuing 24-hour cycle, which gave the least error in terms of determined flow and load optimisation criteria. Mather (1997) studied the use of fuzzy logic for equalization tank control, but concluded that this type of controller would not be adaptive and self-correcting and thus unsuitable for equalization control. Fuzzy logic makes use of linguistic variables. Instead, Mather (1997) developed a control strategy based on iterative mathematical calculations, which is still in use at least in four plants in Johannesburg, South Africa. The control operates within the following constraints:

- The equalization volume available
- The equalization basin should neither overflow nor be drawn down to a level below the minimum
- The inverse relationship between volumetric flow and pollutant load
- Daily and seasonal variations in the inflow pattern

3.1.5 Beneficial characteristics of the BNR feed water

The suitability of the wastewater for biological treatment can be assessed in several ways. The important aspect of organic matter characterization is the fractionation due to its rate of degradation (Henze, 1992). The rate of degradation determines how readily the organic matter can be used by different bacteria. Nevertheless, different views on the importance of different fractions still exist. The carbon-to-nutrient ratio is also often used as a simple method for assessing the successfulness of the BNR process. Less attention has been paid to the flow and load variation at the plant inlet, and no widely accepted methodology for its assessment has been developed.

3.1.5.1 Readily biodegradable organic matter

The biodegradability of wastewater is usually estimated with bioassays under oxic conditions, such as RBCOD (Wentzel et al., 1995). The readily biodegradable organic matter constitutes usually 10-15% of the raw wastewater total COD. Small molecules, such as VFAs, carbohydrates, alcohols, peptones and amino acids belong to this group, and they are metabolized at a high rate under aerobic and anoxic conditions. Sollfrank and Gujer (1991) measured the RBCOD from 23 different wastewaters in Switzerland, and the concentration corresponded to 15.5% (±1.8%) of the total COD. Vollertsen et al. (2006) reported low VFA concentrations in the influent wastewater in Denmark. Rapidly hydrolysable organic matter is in soluble form and has varying hydrolysis rates, whereas slowly hydrolysable organic matter is in suspended form and has an even wider range of hydrolysis rates (Orhon et al., 1999). The rapidly hydrolysable fraction represents about 15–20% of the total COD, and the slowly hydrolysable fraction accounts for 40–60% (Henze, 1992)

The use of oxic bioassays for estimating the wastewater's suitability for denitrification has been criticised because microbial availability of organic matter is not the same under oxic and anoxic conditions. Alternative bioassays in anoxic conditions have also been proposed, e.g. a method using 45 days anoxic incubation with excess nitrate (Tusseau-Vuillemin et al., 2003). Several methods have also been proposed to measure the VFA forming potential (Lie and Welander, 1997; Ruel et al., 2002). Recently, the characterization of nitrogen compounds has also been researched because in order to reach very low effluent total nitrogen concentrations, the non-biodegradable part of the nitrogen becomes relevant (Sharp et al., 2009; Sattayatewa et al., 2009).

3.1.5.2 Nutrient-to-carbon ratios

Early experiences with biological nutrient removal showed that the total Kjeldahl nitrogen (TKN) to COD ratio affects denitrification and the total phosphorus (TP) to COD ratio influences the enhanced biological phosphorus removal. According to

Pitman (1990), the TP:COD ratio in the feed water should not exceed 0.02 if phosphorus removal is to be completed without simultaneous chemical precipitation. More recently, Barnard and Abraham (2006) stated that a TP:COD ratio lower than 0,025 could be used as a rough estimate of the ability of a plant to remove phosphorus, although no distinction was made to the form of the available carbon. The theoretical COD demand can be calculated from the stoichiometric relationship, but in process conditions the ratio should always be higher. The COD demand depends on the carbon source (Swinarski et al. 2009). About 8.6 mg COD was needed to reduce 1 mg NO_3-N to nitrogen gas during denitrification and about 50 mg COD, or 7 to 9 mg VFA (Cuevas-Rodrigues et al., 1998), was required per 1 mg of total phosphorus removed (Rössle and Pretorius, 2001). Æsøy and Ødegaard (1994) reported that 6 g COD/ g NO_3-N would be consumed. Rössle and Pretorius (2001) collected characteristic ratios from several countries – the TKN:COD ratio was typically between 0.06 and 0.10, and the TP:COD ratio was between 0.015 and 0.028. As a matter of fact, the nutrient-to-carbon ratios are only a different way of assessing whether the readily biodegradable fraction of organic matter is sufficient. Jönsson *et al.* (1996) established relationships between VFA potential and total COD, soluble COD and BOD, but Laitala (2005) observed that total COD did not give much information about the available organic substrate.

3.1.6 Performance of the multi-purpose pre-treatment at the Pihlajaniemi WWTP

As described in detail in Section 2.5, one pair of existing primary clarifiers was used for:

- diurnal flow equalization with in-line configuration
- in-line prefermentation (APT) with sludge recycle to the middle of the basin
- removal of primary solids with existing scrapers

The most important data concerning the performance of the multipurpose pre-treatment are presented and discussed in Appendix I.

3.1.6.1 Flow equalization

Since no widely accepted parameters exist for measuring the efficiency of the flow equalization, the successfulness of the flow equalization was measured with a coefficient created for this purpose. The coefficient was called the flow variation coefficient, and it enabled the unifying of the variability for different flow rates. (II) The coefficient gives a higher value when the variations are higher. The coefficient was calculated by dividing the standard deviation of flow rates from the previous 24 hours with the mean value of the same flow rates:

Flow variation coefficient = standard deviation$_{q,prev\ 24h}$ / mean value $_{q,prev\ 24h}$ (4)

As can be seen from the flow variation coefficients presented in Figure 3.3, flow variations were decreased in the equalization basin during the whole study. Flow equalization was particularly successful during February–March 2003, June–September 2003, and January-March 2004. These periods corresponded to dry weather periods. Overall, the equalization gave satisfactory results when the influent daily average flow was less than 1.3 times the long-term daily average flow. As expected, the equalization basin volume was not sufficient to store the water nor was the retention time sufficient for hydrolysis during the peak flows (Appendix I). During the first months, equalization was unsatisfactory due to the tuning of the control of the equalization pump. It can be seen as well that the flow variations were considerably levelled out also in the primary clarifier process train compared to the influent flow. This was due to reject water recycles at the plant and the bypass flow of the primary clarification train of around 20%. Moreover, some flow equalization always occurs at each step of the process.

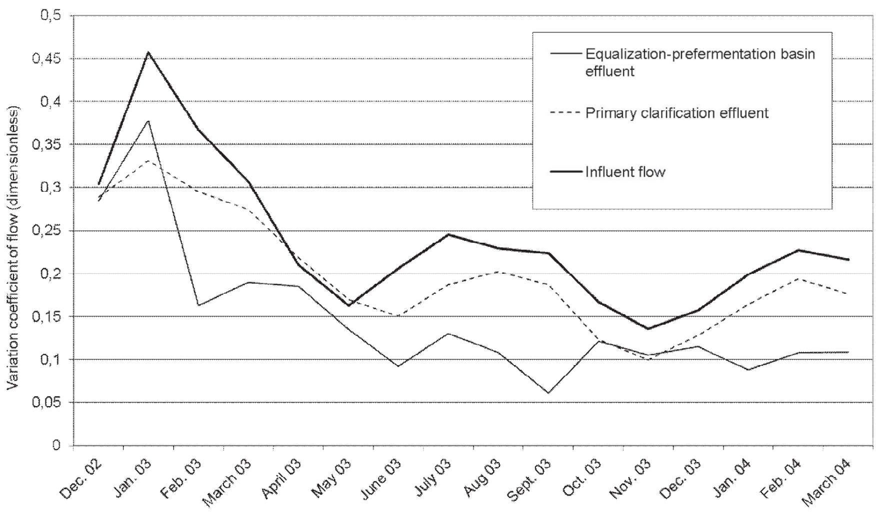

Figure 3.3 Flow variation coefficients from the influent flow and from equalization /prefermentation and primary clarification process trains.

3.1.6.2 Removal of solids, hydrolysis and nutrient solubilisation

The concentrations of different parameters measured in the equalization/prefermentation effluent and in the primary clarification effluent are presented in tables 3.2 and 3.3. In order to assess the differences between the two pre-treatment effluents as well as between plant influent wastewater and pre-treated wastewater, statistical analysis with a *t*-test was used. The results of the statistical analysis are also shown in tables 3.2 and 3.3. It can be seen that the equalization/prefermentation basin was as efficient as the primary clarifier in the removal of particulate matter. As a matter of fact, the removal rates of total COD, total BOD, and suspended solids were higher in the equalization/prefermentation basin than in the primary clarification basin during test periods 1 and 2. When the internal sludge recycle was added (test periods 3 and 4), both basins achieved similar removal rates. Anyway, it should be stressed that the primary clarification basin was partly bypassed and thus the removal of solids could have been less efficient than in conventional primary clarification. Nevertheless, on average, BOD and COD were removed by 33% and 38%, respectively, in both process trains, whereas suspended solids decreased by 60% in the equalization/prefermentation process train and 55% in the primary clarifier process train. These removal rates are very close to typical values presented in the literature (Harremoës et al., 1993).

Soluble organic matter, on the other hand, increased significantly in the equalization/prefermentation basin, suggesting that organic matter was hydrolysed in the basin. The hydrolysis took place already during periods 1 and 2, when the basin was operated as equalization basin only. The VFA yield was relatively low, which anyway was well within range of reported results in the literature (Appendix II; Rössle and Pretorius, 2001). The concentration of soluble COD and BOD was also higher in the primary clarifier effluent than in the plant influent, which supported the findings of Gemaye et al, (2001). Nevertheless, the equalization/prefermentation effluent soluble BOD concentration overall was 12% higher and the soluble COD concentrations were 18% higher on average than in the primary clarifier effluent during the whole study. The retention time in the prefermenter significantly increased the soluble organic matter (I, II), and as expected, the proportion of soluble organic matter leaving the equalization/prefermentation basin. The effect of temperature was not pronounced and no significant correlation was found, although concentrations of soluble organic matter were higher during the two test periods (2 and 3) in summertime. The results presented in the literature suggest that prefermentation was enhanced in warmer temperature (Wedi, 1992; Christensson et al., 1998; Teichgräber, 2000; Chanona et al., 2006). The less pronounced effect of temperature could be attributed to the fact that during summertime, soluble organic matter was also consumed more rapidly in the sewer network. No clear improvement in the elutriation of organic matter was observed when a sludge recycle was added and sludge removal was decreased, increasing the SRT in the basin (test periods 3 and 4). Interestingly, no significant difference in VFA concentration could be seen between primary clarification and equalization/prefermentation basin, although soluble COD and BOD concentrations were approximately 15% higher in the prefermented water and RBCOD concentration was also significantly higher (I). This could be due to the fact that more VFA was consumed or evaporated due to more turbulent flow at the pump outlet in the equalization/prefermentation process train. The daily average values of VFA varied between 4 mg/l and 34 mg/l in both process trains.

The pH value increased in the pre-treatment from 7–7.6 to around 8.0, although the production of VFAs should be accompanied by a small decrease in pH (Bajaras et al., 2002; Christensson et al., 1998). The observed increase was attributed to the sludge thickener reject water that was led to the beginning of the sand removal basin. Lime was added to the thickener in order to prevent secondary phosphorus release. Moreover, influent wastewater contained 1.8–6 mg/l of iron. The mass balance calculation revealed that 70–80% of the iron was removed with the raw sludge during wintertime, whereas during summertime a part of the iron was released and only 40-50% was found in the raw sludge. This indicated that the conditions during wintertime in the prefermentation basin were not totally anaerobic.

Nitrogen was solubilised in the equalization/prefermentation basin. The phosphate concentration was slightly higher in the equalization/prefermentation effluent than in the primary clarifier effluent as well, but the difference was not statistically significant. The measured increase in the NH_4-N was 0.04-0.16 mg NH_4-N/mg RBCOD and 0.004 – 0.016 mg NH_4-N/mg influent COD. The values are in the range of values (0.002 – 0.007 mg NH_4-N/mg influent COD) stated in the literature (Barajas et al., 2002). However, it was also observed based on the loads in both process trains that the solubilisation of nutrients in the prefermenter did not increase the total load to the biological process (I). This would mean that mainly compounds that would not be removed in the primary clarification were solubilised. It should be also noted that in Pihlajaniemi, reject waters from sludge dewatering and wash water from the tertiary filters were added to the process before pre-treatment, and thus some of the measured increase in ammonium and phosphorus could have been caused by the reject waters. Furthermore, the proportion of ammonium of the total nitrogen did not show any correlation with the retention time, suggesting again that an important part of the ammonium increase was coming from the reject water or also that proteins, which are the main source for the released ammonium, degrade first to amino acids and further to ammonium fairly rapidly (Teichgräber, 2000).

Table 3.2 Concentrations in the plant influent, after equalization/prefermentation and after primary clarification and the significance of difference between the different pre-treatment process trains according to the T-test.

				BOD_7(ATU)	BOD_7(ATU) $_{soluble}$	COD_{Cr}	$COD_{Cr\ soluble}$
				mg/l			
Period 1	cold, no recycle	21.11.02-7.4.03	Conc. Influent avg	265	84	769	180
			Conc. equal avg	177	98	433	223
			Conc. clar. Avg	186	86	468	205
			% diff equal/clar	-5%	+14%	-8%	+8%
			T-test influent/equal	0.000*	-	0.000*	-
			T-test influent/clar	0.000*	-	0.000*	-
			T-test equal/clar	0.404	0.061	0.137	0.129
Period 2	warm, no recycle	1.6.03-26.8.03	Conc. Influent avg	164	26	495	104
			Conc. equal avg	106	49	319	137
			Conc. clar. Avg	112	37	348	106
			% diff equal/clar	-6%	+33%	-8%	+30%
			T-test influent/equal	0.000*	0.025*	0.000*	0.027*
			T-test influent/clar	0.000*	0.274	0.000*	0.807
			T-test equal/clar	0.610	0.170	0.360	0.002*
Period 3	warm, with recycle	26.8.03-16.10.03	Conc. Influent avg	177	34	547	110
			Conc. equal avg	121	53	374	164
			Conc. clar. Avg	112	39	336	140
			% diff equal/clar	+8%	+37%	+11%	+18%
			T-test influent/equal	0.019*	-	0.002*	-
			T-test influent/clar	0.008*	-	0.000*	-
			T-test equal/clar	0.660	0.083	0.368	0.167
Period 4	cold, with recycle	1.1.04-15.3.04	Conc. Influent avg	207	-	579	-
			Conc. equal avg	147	61	381	169
			Conc. clar. Avg	146	56	395	162
			% diff equal/clar	0%	+8%	-4%	+4%
			T-test influent/equal	0.000*	-	0.000*	-
			T-test influent/clar	0.000*	-	0.000*	-
			T-test equal/clar	0.906	0.299	0.631	0.499
Whole study		21.11.02-15.3.04	Conc. Influent avg	200	35	582	115
			Conc. equal avg	135	63	360	164
			Conc. clar. Avg	133	53	363	147
			% diff equal/clar	0%	+18%	-1%	+12%
			T-test influent/equal	0.000*	0.017*	0.000*	0.015*
			T-test influent/clar	0.000*	0.096	0.000*	0.107
			T-test equal/clar	0.864	0.048*	0.828	0.048*

* Statistically significant difference

Table 3.3 Concentrations in the plant influent after equalization/prefermentation and primary clarification and the significance of difference between the different pre-treatment process trains according to the T-test.

				N_{tot}	NH_4-N	P_{tot}	PO_4-P	Suspended solids
				mg/l				
Period 1	cold, no recycle	21.11.02-7.4.03	Conc. influent avg	66	52	12	6	380
			Conc. equal avg	59	55	9	6	152
			Conc. clar. Avg	57	50	10	6	188
			% diff equal/clar	+3%	+10%	-5%	+7%	-19%
			T-test influent/equal	0.001*	0.170	0.000*	-	0.000*
			T-test influent/clar	0.000*	0452	0.000*	-	0.000*
			T-test equal/clar	0.381	0.006*	0.181	0.138	0.079
Period 2	warm, no recycle	1.6.03-26.8.03	Conc. influent avg	44	33	9	-	347
			Conc. equal avg	42	41	7	4	124
			Conc. clar. Avg	40	36	8	4	177
			% diff equal/clar	+3%	+14%	-8%	+6%	-30%
			T-test influent/equal	0.408	0.012*	0.003*	-	0.000*
			T-test influent/clar	0.182	0.274	0.015*	-	0.000*
			T-test equal/clar	0.622	0.053	0.243	0.396	0.016*
Period 3	warm, with recycle	26.8.03-16.10.03	Conc. influent avg	43	34	9	4	331
			Conc. equal avg	42	39	8	4	144
			Conc. clar. Avg	39	34	7	4	134
			% diff equal/clar	+9%	+15%	+6%	+4%	+7%
			T-test influent/equal	0.843	0.377	0.104	-	0.000*
			T-test influent/clar	0.435	0.990	0.048*	-	0.000*
			T-test equal/clar	0.622	0.348	0.599	0.752	0.681
Period 4	cold, with recycle	1.1.04-15.3.04	Conc. influent avg	59	49	10	-	340
			Conc. equal avg	54	45	9	4	148
			Conc. clar. Avg	56	43	8	4	163
			% diff equal/clar	-4%	+7%	+2%	+12%	-9%
			T-test influent/equal	0.471	0.210	0.446	-	0.000*
			T-test influent/clar	0.460	0.043*	0.358	-	0.000*
			T-test equal/clar	0.643	0.134	0.873	0.274	0.447
Whole study		21.11.02-15.3.04	Conc. influent avg	50	38	10	5	341
			Conc. equal avg	47	43	8	5	137
			Conc. clar. Avg	45	39	8	4	155
			% diff equal/clar	+4%	+11%	-1%	+9%	-11%
			T-test influent/equal	0.241	0.025*	0.000*	0.619	0.000*
			T-test influent/clar	0.070	0.793	0.000*	0.362	0.000*
			T-test equal/clar	0.437	0.013*	0.850	0.113	0.063

* Statistically significant difference

3.1.6.3 Performance of the pre-treatment from the point of view of BNR

The carbon-to-nutrient ratio, which in the plant influent was fairly advantageous for nutrient removal, deteriorated in both pre-treatment processes. In both process trains, the COD_{Cr}:N_{tot} ratio dropped from 12 to 8, and the COD_{Cr}:P_{tot} ratio dropped from 61 to 47 on average, although the equalization/prefermentation effluent contained significantly more available organic matter. The comparison shows quite clearly that total organic matter does not give any good estimation of the suitability of the wastewater for BNR. On the contrary, the soluble organic matter increased, as explained in the previous chapter. RBCOD and VFA concentrations also increased (II, V).

The VFA produced in the pre-treatment in both process trains consisted mainly of acetic acid and propionic acid. Acetic acid was produced predominantly (80–99%). Propionic acid accounted for 1-14% of the VFAs, and butyric and valeric acids were rarely observed. The VFA-COD equivalent in the equalization/prefermentation process train was 1.11, whereas in the clarification process train it was 1.14. These values are close to the value (1.15) of the COD equivalent in the literature for APT applications (Münch and Koch, 1999). The higher VFA-COD equivalent observed for the primary clarifier process train could be because the sludge blanket was denser in the primary clarification process train, whereas the sludge recycle and pumping resulted in a less concentrated sludge blanket.

The production rate of soluble COD during the pre-treatment is presented in Figure 3.4. The production rate was clearly higher in the equalization/prefermentation (7 mg/l/h) basin than in the clarifier (2 mg/l/h). It can also be seen that during high flow rates, the soluble COD production dropped in both process trains. The production rates of VFAs are presented in Appendix I.

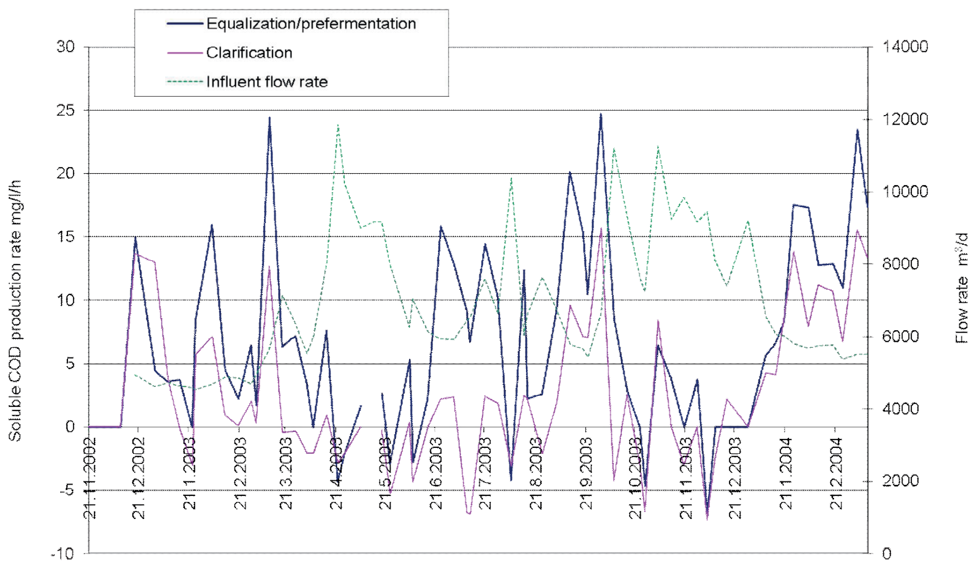

Figure 3.4 Soluble COD production rate in the equalization/prefermentation and in the primary clarification and the influent flow rate during the whole study.

3.2 The effects of equalization and prefermentation on biological nutrient removal

3.2.1 Introduction

In the following, a short description of the present understanding of the nutrient removal processes is given with the focus on potential effects of equalization and prefermentation on their performance. In the end, the results from the Pihlajaniemi WWTP are presented. The conventional activated sludge process aiming only at removal of organic matter is not included because its requirements are usually not limiting the process performance.

3.2.2 Nitrification

3.2.2.1 Biochemical background of nitrification

Nitrification is the conversion of ammonium to nitrate by microbial activities. It can occur in aerobic conditions by autotrophic or heterotrophic bacteria (usually fungi). Ammonium can also be oxidized to form N_2 by autotrophic bacteria by using nitrite as the electron acceptor (also called Anammox). (Lo, 2008). In the activated sludge process conditions, nitrification is aerobic and autotrophic and has two steps. Bacteria, such as *Nitrosomonas, Nitrospira, Nitrosococcus* and *Nitrosolobus,* oxidize ammonium to nitrite and are collectively termed ammonium oxidizing bacteria (AOB). Nitrite is then converted to nitrate by *Nitrobacter* or *Nitrospira, collectively named nitrite oxidizing bacteria (NOB).* Approximate equations for the reactions that occur can be written as:

For *Nitrosomonas*:

$$55\ NH_4^+ + 76\ O_2 + HCO_3^- \rightarrow C_5H_7O_2N + 54\ NO_2^- + 57\ H_2O + 104\ H_2CO_3$$

For *Nitrobacter*:

$$400\ NO_2^- + NH_4^- + 4\ H_2CO_3 + HCO_3^-\ 195\ O_2 \rightarrow C_5H_7O_2N + 3\ H_2O + 400\ NO_3^-$$

It can be noted that approximately 4.3 mg of O_2 is required to oxidize 1 mg of ammonium-nitrogen to nitrate-nitrogen. In the conversion process, a large amount of alkalinity is consumed: 8.64 mg HCO_3^- per mg of ammonium-nitrogen oxidized (Metcalf and Eddy, 1991, p. 431). In common biological wastewater treatment, the nitrifying biomass is only a small fraction, 1-5%, of the total biomass (Harremoës et al., 1993).

3.2.2.2 Factors influencing nitrification

The influence of temperature on nitrification is well-established. The temperature coefficient θ in Arrhenius equation which provides a generalized estimate of temperature effects on biological reaction rates, varies between0.08 and 0.13 $°C^{-1}$ for nitrification (Baetens et al., 1999, Head and Oleszkiewicz, 2004). Each biomass, including AOB and NOB, has their own temperature dependency factors. Moreover, the growth rates of AOB and NOB are affected by the substrate (ammonium and nitrite) concentration, BOD/TKN ratio, DO concentration, alkalinity and pH (Metcalf and Eddy, 1991, p. 700; Ekama et al., 1984). The decreased nitrification rate in cold temperature can be compensated for by a longer sludge age (Sinkjær et al., 1994; Henze et al., 2000; Komorowska-Kaufman et al., 2006) Nitrification, together with secondary sedimentation, is considered the limiting process of the biological wastewater treatment in the dynamic conditions at WWTP and during the storm water period (Tränckner et al., 2007). The nitrifying bacteria are growing more slowly than the heterotrophs and are also more sensitive to sudden changes in flow, temperature, etc. (Harremoës et al., 1993; Müller and Krauth, 1998; Le Tallec et al., 1997). The maximum specific growth rate of nitrifying bacteria has been observed to be specific to the wastewater source caused by inhibitory substances in the wastewater. Dytczak et al. (2007) observed significantly higher nitrification rates in alternating anoxic-aerobic conditions than in only aerobic conditions. The authors could rule out the effect of pH and alkalinity, and they concluded that the difference was due to a different bacterial population. High oxygen and substrate levels as well as high nitrite levels would have favoured the *Nitrosomonas* and *Nitrobacter* species having higher ammonium and nitrite oxidation rates over *Nitrosospira* and *Nitrospira*.

3.2.2.3 Effects of equalization and prefermentation on the nitrification process

Only a few published studies were available on the possible effects of flow equalization and prefermentation on nitrification. The reported effects of diurnal flow equalization are mainly positive, whereas storage tanks might lead to negative impacts. LeTallec et al. (1997) showed in a fixed-film process that the filter fed with a low variation of load led to a reduced outlet NH_4-N concentration compared with the filter with normal variation in influent load. The low variation meant applied ammonium load varying between 0.2 and 1.15 kg NH_4-N/m^3/d, while the normal variation was 0.1–1.5 NH_4-N/m^3/d. The authors observed that the amount of efficient autotrophic bacteria was higher in the filter with low load variation. Bolmstedt and Olsson (2005) observed by equalization basin simulation that during dry weather the plant effluent ammonium peaks of 8 mg/l could be decreased to 3 mg/l. The increased nitrification did not increase the aeration cost; rather, the absence of flow rate peaks allowed for better utilization of the available oxygen. On the other hand, Rosenwinkel et al. (2007) demonstrated by simulation and with full-scale plant data that nitrogen removal could handle flow rates up to two times the normal dry weather flow, but with a quadruple increase in the hydraulic load the effluent standards could not always be met. Moreover, Funamizu et al. (2000) observed in a laboratory-scale test that nitrogen removal was less affected by a short-term pulse-shaped increase of load than with a longer period of low organic and nitrogen load. This observation was supported by the findings of Jack and Ashley (2002) with a continual decline in nitrifying biomass concentrations due to the dilute influent feeding from the storage tank. These results suggest that the use of a storage tank might be detrimental for the recovery of the nitrification as it causes a longer period of dilute loading. Bertrand-Krajewski et al. (1995) proposed that the increase in the mineral solids fraction in the influent wastewater after storm events could have a long-term (several days) negative impact on biological activity of sludge due to a decrease in the VSS/TSS ratio.

The reported studies on the effects of prefermentation are very few and somewhat confusing. McCue et al. (2003) observed improved nitrification rates and ammonium removal in a process train receiving prefermented wastewater compared with a process train receiving pre-settled wastewater. The authors could not explain the improvement. A possible explanation based on the findings of this research is presented in Appendix II. Takai et al. (1997), on the other hand, observed that VFA could inhibit nitrification. Nitrite oxidation was more sensitive to VFA than ammonium oxidation.

3.2.3 Denitrification

3.2.3.1 Biochemical background of denitrification

The conversion of nitrate-nitrogen to nitrogen gas can be accomplished by several genera of bacteria. In wastewater treatment, the process is mainly accomplished by heterotrophic bacteria capable of dissimilatory nitrate reduction in two steps. The first step is conversion of nitrate to nitrite. This step is followed by production of nitric oxide, nitrous oxide and finally nitrogen gas. (Metcalf and Eddy, 1991, p. 432). However, denitrification also takes place under autotrophic as well as autotrophic and heterotrophic mixed conditions (Lo, 2008). It has been shown that some polyphosphate-accumulating organisms (PAOs) are able to denitrify (Ekama and Wentzel, 1999). In this case, the denitrifying PAOs will utilize nitrate as the electron acceptor and will consume the same amount of organic substrate for denitrification and phosphate uptake simultaneously (You et al., 2001). The non-denitrifying PAOs can only utilize oxygen as electron acceptor.

3.2.3.2 Factors affecting denitrification

The denitrification rate is affected by nitrate concentration, carbon concentration, the type of carbon source, temperature, dissolved oxygen and pH (Metcalf and Eddy, 1991, p. 723). The heterotrophic denitrification requires organic carbon as a carbon source for cell synthesis and an electron donor for energy production. In the pre-denitrification configuration, wastewater's own carbon content can be used, but post-denitrification usually requires external carbon. The absence of oxygen is also an important factor because the bacteria will prefer oxygen over nitrate or nitrite. Lie and Welander (1994) found a linear relationship between denitrification rate and ORP. They also pointed out that an oxygen concentration below the detection limit can inhibit denitrification.

You et al. (2001) showed that the specific denitrification rate was higher when soluble phosphates were present. Siegrist and Gujer (1994) stated that denitrification in the secondary clarifier can substantially increase nitrogen removal, but enhanced denitrification in the secondary clarifier is not recommended due to the risk of nitrogen bubbles pushing the sludge to the surface. Siegrist et al. (1995) observed in a full-scale plant that 30% of the total denitrification capacity was in the secondary clarifiers. They proposed a model for denitrification in the secondary clarifiers based on the ratio of influent to return sludge flow and scraper interval.

3.2.3.3 Effects of equalization and prefermentation on the denitrification process

The importance of the suitable organic matter for denitrification has been widely studied, and the dependence of denitrifying bacteria on suitable organic matter has been established (*inter alia* Kristensen et al., 1992; Brinch et al., 1994; Æsøy and Ødegaard, 1994; Hatziconstantinou et al., 1996). Prefermentation units are in many cases added to fulfil the requirements of denitrification. Kristensen et al. (1992) evaluated that biological sludge hydrolysis offers a potential increase of 10-15% in the biological nitrogen removal. Brinch et al. (1994) reported an increase of approximately 44% in the denitrification rate through the dosing of hydrolysate. Takai et al. (1997) observed that acetic acid was the preferred carbon source for denitrifying bacteria over propionic acid. Hallin and Pell (1998) also showed that denitrifying bacteria maintain their ability to denitrify with a feed of several carbon compounds after adaptation to ethanol better than when adapted to methanol. Moser-Engeler et al. (1998) observed a higher denitrification rate with a mixture of six fermentation products than with single substrates acetate and propionate. Komorovska-Kaufman et al. (2006) showed that the denitrification rate increased with the addition of VFA up to the level 1.67 mg CH_3COOH/mg N and after that stabilized. Swinarski et al. (2009) studied the suitability of wastewater from different food industries as an alternative carbon source for denitrification. They showed that distillery, brewery and fish-pickling wastewater improved denitrification efficiency, although the nitrogen mass balance calculation revealed that the nitrate recycle rate had a significantly higher influence on the effluent NO_3-N concentration than the denitrification rate.

The effects of equalization on denitrification, on the other hand, have not been studied extensively. Van Loosdrecht et al. (1997) presented a hypothesis that the development of bacterial population capable of storing intracellular polyhydroxyburyrate (PHB) due to a dynamic feed would have a negative effect on denitrification because a part of the COD in the anoxic zone would be stored in the cells as PHB and transferred to the aerobic zone. This could explain why the COD requirements for denitrification are usually higher than the theoretical need. According to Hasselblad and Hallin (1998), the denitrification potential was only affected when the organic matter load was decreased for 5 days. A 24h decrease in the organic feed did not affect the denitrification potential. The denitrification potential was determined in triplicate with the acetylene inhibition technique.

3.2.4 Enhanced biological phosphorus removal

3.2.4.1 Biochemical background of enhanced biological phosphorus removal

The biochemical model of the behaviour of PAOs can be summarized as follows (Wentzel et al., 1991):

- VFA serve as substrate for PAOs.
- In the anaerobic phase, the VFA are taken up and stored as polyhydroxyalkanoates (PHA). This process requires energy, which is supplied by the breakdown of polyP.
- In the aerobic reactor, the stored PHA is used as a carbon and energy source for the cell function and growth and as an energy source for polyP formation which gives rise to P uptake.

- At the same time, the intracellular glycogen is degraded to maintain the redox balance of the cell. Under aerobic conditions, the glycogen pool is regenerated (You et al., 2001).
- The synthesis of poly-P is accompanied by an additional uptake of magnesium and potassium ions, which are the typical counterions in the poly-P synthesis. The function of these cations in the bacterial cells is to neutralise the negative load of the polyphosphates and to activate the poly-P synthesising enzymes (Jardin and Pöpel, 1996).

It seems that the bacteria community taking part in the enhanced biological phosphorus removal (EBPR) process is diverse and consists of several major groups of microorganisms, also including some denitrifiers (Ekama and Wentzel, 1999). It has also been suggested that a wide range of organic matter other than VFA can be utilized by PAOs and storage polymers other than PHA are found (Mino et al., 1998; Carucci et al., 1999). Furthermore, evidence has been found that other groups of microorganisms are capable of storing VFA without performing EBPR (e.g. glycogen-accumulating organisms, GAOs)- Their presence in a enhanced biological phosphorus removal process is undesirable because they enter in competition for RBCOD with PAOs (Vollertsen et al., 2006). On the other hand, recently evidence has been found suggesting that PAOs and GAOs would actually be the same organisms and in certain conditions (such as temperatures above 20 °C, low COD/P ratio and low pH), a metabolic shift towards using glycogen would occur (Erdal et al., 2008; Schuler and Jenkins, 2003).

3.2.4.2 Factors influencing the enhanced biological phosphorus removal

Temperature has a complex influence on phosphorus removal. Usually, the phosphorus release in the anaerobic phase increases with increasing temperature. It has been shown that temperature affects the process kinetics but has no influence on stoichiometry. In the aerobic phase of EBPR, there are four independent rate temperature coefficients (growth, P-uptake, glycogen formation, and maintenance), which have complex interactions between them. The influence of temperature on phosphorus removal is difficult to predict because of the conflicting effects of sub-processes – e.g. process kinetics will decrease in lower temperatures, but PAOs' decay will decrease as well. Temperature will also affect nitrification and thus denitrification leaving more substrate to PAOs. If simultaneous precipitation is used, the formation of precipitates will also be affected by temperature. Furthermore, the hydrolysis rate, which is also temperature-dependent, affects the P release (Baetens et al., 1999; Vollertsen et al., 2006). Moreover, it has been observed that composition of the bacterial population shifted with temperature (5–30°C). (Brdjanovic et al., 1997; 1998b).

In addition to temperature, the ratio between phosphate release and substrate uptake is a significant factor for the EBPR process, e.g. if PAOs take up a substrate with a high $\Delta P/\Delta S$ ratio, they accumulate a large amount of polyphosphate in the anaerobic phase. (Moser-Engeler et al., 1998; Schuler and Jenkins, 2003). The P release to substrate uptake ratio has been reported to depend on pH (Filipe et al., 2001), the ratio between COD and P in the feed water, and the extent of dominance of PAOs over GAOs in the sludge (Rodrigo *et al*, 1999; Schuler and Jenkins, 2003). Moreover, phosphorus removal is affected by nitrate, nitrite, calcium, magnesium, potassium, HRT, SRT and the conditions in the aerobic zone. If nitrates enter the anaerobic zone, the readily biodegradable organic matter will be consumed preferentially by denitrifiers (Pitman et al., 1988). Nitrates will also prevent development of the anaerobic conditions necessary for VFA formation (Barnard and Abraham, 2006). The authors also pointed out that if the RBCOD is converted to VFA beforehand, the inhibitory effect of nitrates on phosphorus removal will be less pronounced. Saito et al. (2007) reported that also nitrite could have an inhibitory effect, but its importance would depend on the aerobic nitrite denitrification activity of PAOs. Yoshida et al. (2006) showed that in an A/O process, inhibition started already with 1 mgN/l of initial nitrite. Furthermore, nitrite inhibition continued after nitrite disappearance and that PAOs with higher relative anoxic activity were less sensitive to nitrite exposure. Yoshida et al. (2009) studied the nitrite inhibition further and proposed that the inhibiting mechanisms of nitrite would be based on the reaction of either nitrite or produced nitric oxide with oxygen respiration reductase, inhibiting the oxygen respiration. Ca, Mg and K are the principal metal components of polyphosphate granules within the cell (de-Bashan and Bashan, 2004). Schoenborn et al. (2001) observed enhanced phosphorus removal when the concentration of magnesium was doubled in the influent. However, Barat et al. (2006) observed that in high calcium concentrations, polyP granules are formed with calcium as a counterion, making them unavailable for the EBPR process. Rodrigo et al. (1996) showed that the phosphorus release and uptake were affected by the sludge age. Overall, increased sludge age (30–60 days) was found to decrease the phosphorus removal. However, adequate retention time is required for the enrichment of PAOs. PAOs facing long aerobic periods may not need the polyphosphate accumulation mechanism to survive short anaerobic periods (de-Bashan and Bashan, 2004). Narayanan et al. (2007) observed that sometimes the aerobic phosphorus uptake can limit the EBPR process. They concluded that a strong initial uptake in the beginning of the aerated zone is essential because the conditions in the initial aerobic zone (high intracellular PHA concentration and high ortho-P concentration) are conducive to a good P uptake. It is important therefore to ensure that DO or other conditions are not limiting the P uptake in the beginning of the aerated zone.

Adding precipitation chemicals to the activated sludge in EBPR systems is possible. Lötter and Pitman (1992) reported that the addition of VFA-enriched feed significantly reduced the chemical dose required to achieve the phosphate standard. According to the authors, the ratio Fe/P in a purely chemical treatment would normally range between 2.0 and 2.5, the stoichiometric ratio being 1.8. Ratios below this would give an indication of the level of biological treatment. According to Maurer and Boller (1999), the chemical fixation is faster and 'stronger', although mostly reversible. Thus, an extensive use of precipitation chemicals is able to suppress the EBPR ability on the sludge. The authors suggest that the amount of dosed chemical should be minimized and the dosing should take place as late as possible. On the other hand, Maurer and Boller (1999) described biologically induced phosphorus precipitation, which can occur in conditions with high phosphate concentration, relatively high Ca concentration, and pH between 7 and 8. Such conditions are typical for the P release of anaerobic zone in the EBPR process and may lead to the formation of apatite. The biologically induced precipitation will not have negative effects on biological phosphorus removal, but on the contrary will improve the phosphorus removal.

3.2.4.3 The effect of equalization and prefermentation on enhanced biological phosphorus removal

Enhanced biological phosphorus removal is often limited by sufficient organic carbon. The VFA are usually considered as the main substrate for PAOs (*inter alia* Wentzel et al., 1991; Lötter and Pitman, 1992; McCue et al., 2003). Thus, the positive effect of prefermentation on EBPR is marked. Wentzel et al. (1991) concluded their previous findings on the enhanced biological phosphorus removal by stating that the mass of VFA that becomes available in the anaerobic reactor is controlling the rate of VFA sequestration and thus phosphorus release. The first observations from BNR plants showed that when RBCOD or VFA concentration in the feed water was above 100 mg/l, enhanced biological phosphorus removal was easy, whereas with a feed water concentration below 50 mg/l, special care should be taken in the process design (Pitman, 1990). It has been suggested that the 14 to 20 mg COD equivalent of acetic acid or VFA potential is needed for the removal of 1 mg of phosphorus (Abu-ghararah and Randall, 1991; Jönsson et al., 1996). Moser-Engeler et al. (1998) observed that acetate and propionate were taken up very quickly, faster than the branched VFAs. Randall et al. (1997), on the other hand, observed that propionic acid was detrimental to EBPR, whereas all the other C_2-C_5 VFAs were beneficial. Alcohols showed small or negligible effect compared to VFAs. It has been observed that the type of VFA affected the ratio between phosphate release and substrate uptake, with acetic acid and isovaleric acid being the most efficient acid types (Johansson et al., 1996; Abu-ghararah and Randall, 1991).

The debate on the most suitable feed water to EBPR is still on-going. The researchers are not unanimous on whether VFA should be in the feed water of the biological phosphorus removal plant or whether RBCOD would be enough and additional VFA could be formed by fermentative bacteria in the anaerobic zone of the process (Wentzel et al., 1985; Randall et al., 1994; Carucci et al., 1999b; Jeon and Park, 2000). Nevertheless, it seems that more emphasis should be placed on measuring the RBCOD in the feed water of the anaerobic zone and less on the actual VFA.

Researchers have different views on the effects of diurnal load variations on P release and uptake. Ekama et al. (1984) stated that the mean daily P removal was not affected by the cyclic flow and load conditions. On the other hand, De Lucas Martinez et al. (2001) observed a rapid response of P release to the changes in organic composition in the feed water. The researchers are more unanimous on the impact of longer famine periods (Brdjanovic et al., 1998a). It has been observed that periods of low influent COD on the order of hours to days, e.g. during weekends and holidays, reduce the P release and the production of the internal PHA and glycogen. When carbon supply increased again, P release recovered rapidly, but it lasted 1-2 days before the P uptake returned to normal values, due to the fact that the PAO had to first regenerate their internal PHA and glycogen pools (Temmink et al., 1996; Carucci et al., 1999a). Lee et al. (1996) observed a decrease in enhanced biological phosphorus removal, which could be attributed to the industrial holiday period and to the prolonged periods of rain.

3.2.5 Results from the Pihlajaniemi WWTP

The performance of the BNR process in the two different process trains is reported and discussed in appendices I, II and III. The more important findings are summarized in the following, and some interesting observations are highlighted.

The overall performance of the plant was compared with the plant performance of the years preceding the study. The results are shown in figures 3.5 and 3.6. During the study, the plant performed better for each parameter measured, although only half of the plant was modified. Especially nitrification was improved: 84% ammonium removal was achieved on average during the study against 77% during the preceding three years. Average BOD removal was 98% against 97%, average COD removal 93% against 91%, average total phosphorus removal 95% against 94%, and finally, average SS removal 99% against 98%.

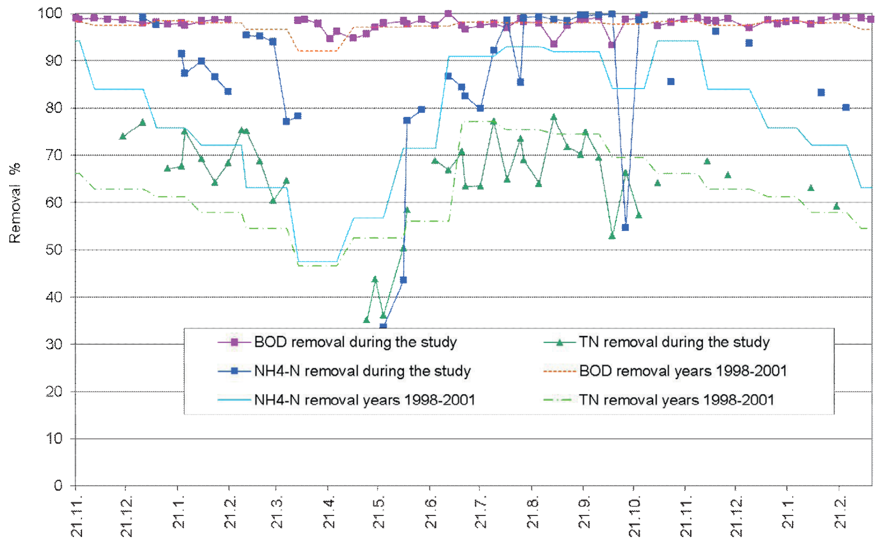

Figure 3.5 BOD, NH_4-N and total nitrogen removal of the whole plant (including the tertiary filtration) during the study and on average during preceding three years.

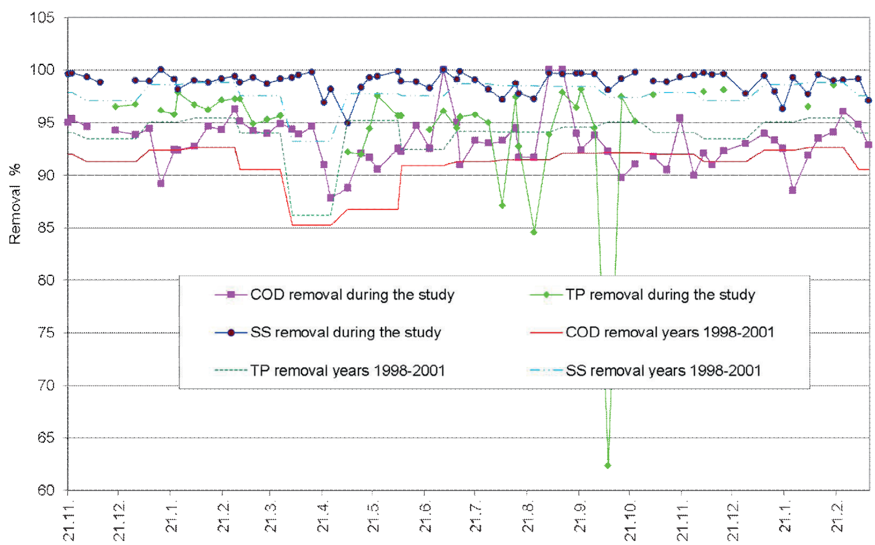

Figure 3.6 COD, SS and total phosphorus removal of the whole plant (including the tertiary filtration) during the study and on average during the three preceding years.

3.2.5.1 Nitrogen removal

Ammonium removal was significantly improved by the modifications in the pre-treatment, as presented in Appendix II. This is also shown in Figure 3.7. The observed improvement in nitrification was attributed to both equalization and prefermentation, although assimilation by heterotrophic bacteria must be taken into account as well. Prefermentation affected directly through ammonification, but also through decreased competition between autotrophic and heterotrophic bacteria. Our findings concerning the effect of prefermentation on nitrification were surprising and novel, and the hypothesis would require verification in a more controlled environment. It is also possible that the conditions, e.g. the nitrite or substrate concentration, favored nitrifying bacteria with a higher oxidation rate in the equalization/prefermentation process train, as proposed by Dytczak et al. (2007). The issue is discussed in detail in II.

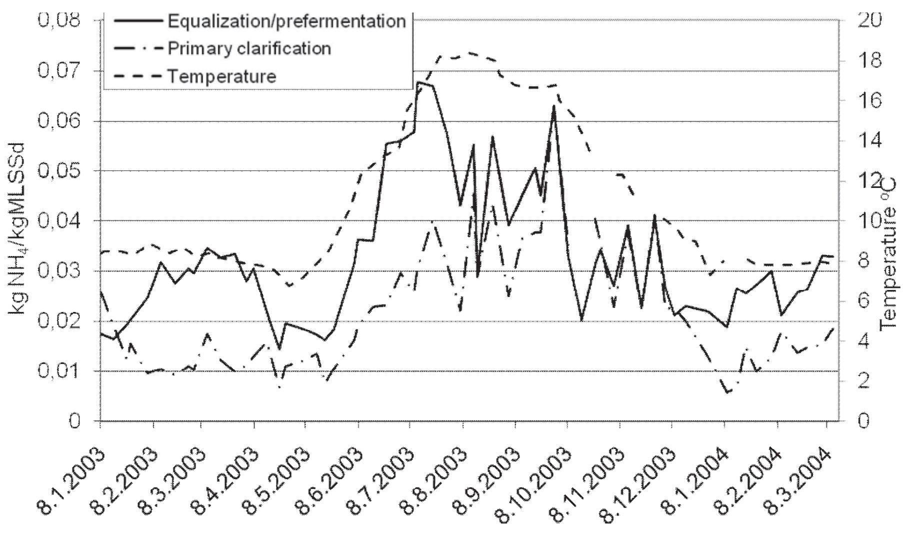

Figure 3.7 Ammonium nitrogen removal rates in the equalization/prefermentation process train and in the primary clarification process train.

The differences in denitrification between the two process trains can be mainly explained by the nitrification, which was the limiting part of the process. A higher amount of nitrate nitrogen was reduced to nitrogen gas in the equalization/prefermentation process train (95 kg/d on average) than in the primary clarification process train (81 kg/d on average), although in the equalization/prefermentation process train, the value represented only 75% of the nitrate nitrogen produced by nitrification, whereas it was 85% in the primary clarification process train. Denitrification presumably suffered in the equalization/prefermentation process train due to the fact that internal recycle flow had to be the same for both process trains. A higher recycle flow rate can significantly improve the denitrification performance (Swinarski et al., 2009). Indeed, a significant correlation was found between the internal recycle rate and the denitrification performance (Pearson's correlation, 0.05) in the equalization/prefermentation process train.

Moreover, denitrification performance in both process trains was hampered by the insufficient quantity of suitable organic matter. The RBCOD produced, in the form of VFA, in the prefermentation was presumably consumed in the anaerobic zone by PAOs. On the other hand, in the anaerobic zone more readily biodegradable material was probably also produced from hydrolysis of slowly biodegradable COD (SBCOD) (Wentzel et al., 1985). To support the fact that suitable organic matter was limiting denitrification, a significant correlation was also found between the soluble COD/total COD ratio and denitrification performance (Pearson's correlation, 0.05).

The role of equalization in denitrification cannot either be completely ruled out, although according to the findings of Haselblad and Hallin (1998), denitrification is not very sensitive for short-term changes in the organic load. On the other hand, a different bacterial population capable of storing organic matter may have developed in the primary clarification process train decreasing the denitrification capacity, as proposed by van Loosdrecht et al. (1998). Also, from time to time, denitrification could have been affected by the oxygen entering the anoxic zone as suggested by Lie and Welander (1994). Unfortunately, the influence of oxygen cannot be clearly estimated because the oxygen concentration (neither the internal recycle nor in the return sludge) was measured on-line.

In our case, denitrification occurring in the secondary clarifiers played an important role in the overall denitrification process because the amount of sludge stored in the secondary clarifiers was substantial, especially in wintertime. This can partly explain why the denitrification performance of the equalization/prefermentation process train was lower during June–August 2003. This period coincides with the period when relatively little sludge was stored in the secondary clarifiers in the equalization/prefermentation process train. On the contrary, the denitrification performance in the primary clarification process train achieved the highest values during the summer months.

3.2.5.2 Phoshorus removal

The results concerning the phosphorus removal are presented and discussed in appendices I and III. Overall, enhanced biological phosphorus removal did not behave quite as expected. In the equalization/prefermentation process train, practically no phosphorus release occurred during the cold weather, whereas during summertime, PO_4-P concentrations up to 42 mg/l in the anaerobic zone were measured. The PO_4-P release rate as a function of temperature in both process trains is shown in Figure 3.8.

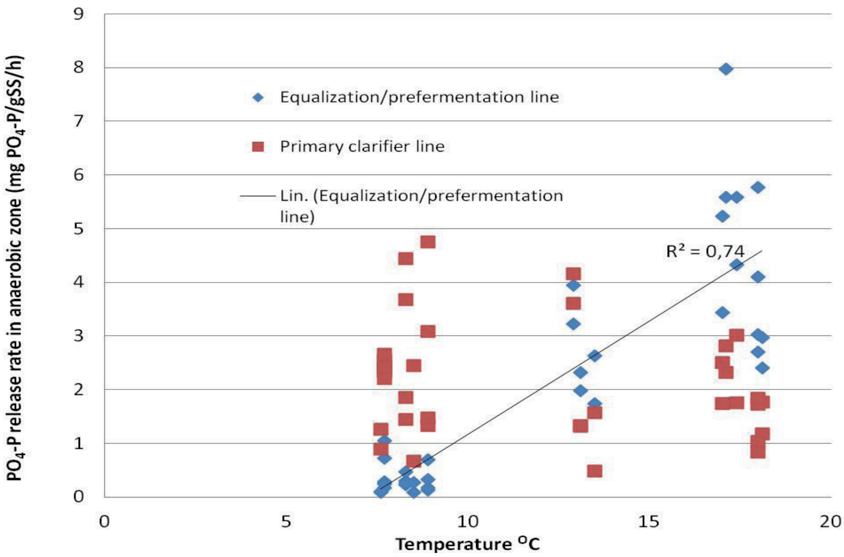

Figure 3.8 Phosphate release rates in an anaerobic zone in the equalization/prefermentation process train and in the primary clarifier process train in relation to wastewater temperature.

The phosphorus release in the equalization/prefermentation process train showed a strong correlation with temperature, supporting the findings of (Vollertsen et al. (2006). Surprisingly, the primary clarification process train behaved very differently from the equalization/prefermentation process train (Figure 3.6). It did not show any correlation with temperature; rather some phosphorus release occurred during the whole study.

Several explanations for the temperature dependence in the equalization/prefermentation process train can be presented: namely, differences in hydrolysis or in VFA feed, influence on bioP process kinetics, aerobic conditions or inhibition by nitrite. During wintertime, VFA and RBCOD concentrations were at the same level as during summertime, suggesting that the temperature would instead have an effect on the bioP process kinetics. On the other hand, the bioP can be quite dependent on the hydrolysis taking place in the anaerobic zone, as suggested by Wentzel et al. (1985). Also, VFA and RBCOD were measured in the effluent of the equalization/prefermentation basin, and VFA could have been consumed in the channel between the basins where at least partly aerobic conditions prevailed. Nitrite was not measured in the return sludge or in the biological reactor. Only nitrate was measured. Thus, nitrite could have been present either in the return sludge or in the biological reactor during wintertime and the PAO population was not able to reduce nitrite, as proposed by Saito et al. (2007). Nitrite could be present in the process as a by-product of nitrification or as an intermediate of denitrification (Yoshida et al., 2005). Indeed, Sin et al. (2007) observed fairly high nitrite values (around 5 mg/l) in the plant effluent when nitrification was not complete. According to Sin et al. (2008) nitrite can play a dominant role, when the process is suffering due to lack of oxygen, low temperature, low sludge age or the presence of inhibitory compounds. Moreover, the anaerobic phase was shortened during wintertime making perhaps the retention time too short, as proposed by de-Bashan and Bashan (2004). One hypothesis would also be that different genera of PAOs would have developed for both process trains. The one in the equalization/prefermentation process train would have been more sensitive to temperature. Acevedo et al. (2012) observed a population shift of PAOs when poly-P availability decreased within days. In our case the differences occurred quite rapidly after the separation and modification of the process trains.

The missing correlation between P release and temperature in the primary clarification process train is difficult to explain. One possible explanation would be that the chemical dosing (which was higher in the primary clarification process train than in the equalization/prefermentation process train due to lower load to the primary clarification process train) would have suppressed the biological activity, and the phosphorus release observed would have resulted from the chemically bound phosphorus. Nevertheless, in both process trains, the Fe/P molar ratio was below the limit for biological phosphorus removal mentioned in the literature (Lötter and Pitman, 1992). The ferrous sulphate was dosed in the beginning of the secondary clarification, but a part of it was recycled back to the anaerobic zone with return sludge. Nevertheless, if this hypothesis were correct, the equalization/prefermentation process train should also have shown at least some similar behavior because both process trains received chemical dosing. The theory of van Loosdrecht *et al.* (1998) when applied to bio P could also explain the difference, and thus the dynamic conditions in the primary clarification process train would have created a bacterial population with the capacity of storing the organic matter and would have suppressed the enhanced biological phosphorus removal. In this case, nevertheless, some correlation with temperature would have been expected. In our case, the limitation due to problems with the aerobic uptake of P, as suggested by Narayanan et al. (2007), is not totally applicable. As a matter of fact, the initial P uptake took place in the anoxic zone (I), suggesting that at least part of the phosphorus accumulating bacteria were denitrifiers (Ekama and Wentzel, 1999). Nevertheless, aerobic conditions were different, as described in the following chapter, and this could have affected the EBPR.

Moreover, the P removal was affected by the performance of the secondary clarifiers. The secondary effluent phosphorus concentration was often higher than the concentration in the aerated zone. It was demonstrated that anaerobic conditions were occurring in the sludge blanket in secondary clarifiers, although the return sludge was aerobic. This is discussed in detail in Appendix III. Occasionally, phosphorus was also escaping with sludge due to hydraulic overload of the secondary clarifiers.

3.2.5.3 Aeration efficiency

In order to assess the aeration efficiency, the Actual Oxygen Requirement (AOR) was calculated as explained in I. Due to the higher load to the equalization/prefermentation process train, the oxygen requirements were also higher. The higher load was mainly due to uneven distribution of the influent wastewater, but the wastewater from the equalization/prefermentation process train was also somewhat more concentrated. The air flow required was also higher in the equalization/prefermentation process train, but the air flow per volume of wastewater treated was the same (I). Nevertheless, it was noticed that the measured air flows from the first zone in the primary clarification process train were too low, and thus the aeration efficiency would be changed a bit in favor of the equalization/prefermentation process train.

The calculated AOR and the measured air flow correlated well in the equalization /prefermentation process train, where the air flow meters were reliable (Figure 3.9). It can also be concluded that in the equalization/prefermentation process train, the aeration efficiency was better because the air flow required for the same volume of more concentrated wastewater was the same or lower than in the primary clarification process train. The improvement was attributed to the constant conditions in the aeration basin, which improved the control of the aeration system. The daily air flow pattern was indeed smoother in the equalization/prefermentation process train (IV), but the hourly DO measurements did not reveal any excessive over-aeration or lack of DO in either of the process trains. Moreover, denitrification was improved by the prefermentation, and this supplied more oxygen to the equalization/prefermentation process train.

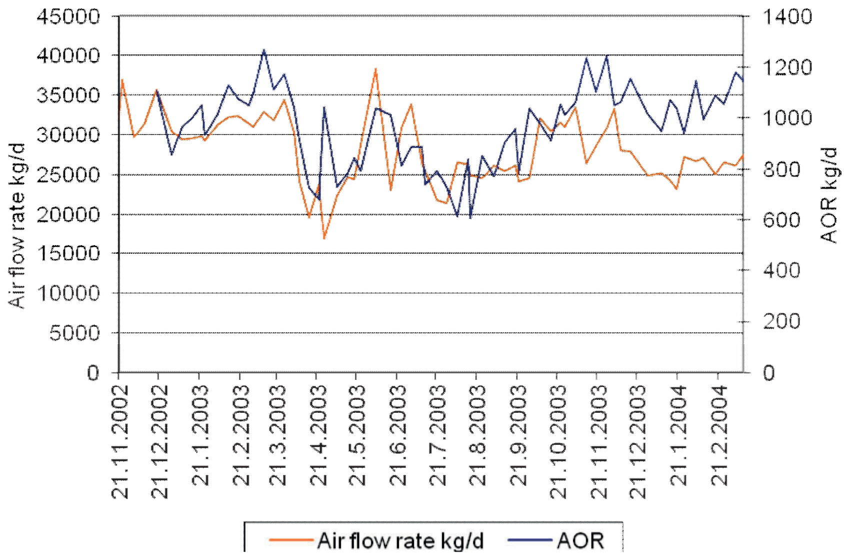

Figure 3.9 The observed air flow and AOR in the equalization/prefermentation process train.

3.3 The effects of equalization and prefermentation on the secondary clarification, sludge characteristics and sludge production

3.3.1 Introduction

Sludge characteristics such as settling and dewatering properties are of high importance because the performance of the secondary clarification determines the successful operation of the activated sludge process. Also, the quantity of biosolids to be transported and processed makes a big difference economically because the cost of sludge treatment and disposal often requires more than 50% of the overall wastewater treatment operational cost (Vesilind and Spinosa, 2001). The more water the biosolids contain, the more expensive the treatment usually is (Kjellén and Andersson, 2002). In the following, secondary clarification and sludge characteristics are discussed with focus on the effects of equalization and prefermentation.

3.3.2 Objective and theoretical background of secondary clarification

The objective of the secondary clarifier is to clarify the overflow and thicken the underflow. Activated sludge flocs are made up of a conglomerate of materials including microorganisms, extracellular polymers, inorganic particulates, non-biodegradable (inert) organic particulates, and water. The settling velocity for each individual floc can be derived from the settling velocity of a spherical particle by adding a term to account for the floc's shape and porosity as well for the effect of flow through the floc. Settling in the dilute suspension is also affected by flocculation (Sears et al., 2006). Sludge flocs with high density have been shown to settle faster than the flocs with low density, although other factors such as shape and size should also be taken into account (Schuler et al., 2001; Sears et al., 2006). On the other hand, sludge that settles too fast cannot produce clear effluent due to the limited flocculation of fine particulates (Yun et al., 2000). At higher concentrations, e.g. in lower parts of the settler, hindered settling followed by transition settling and compression settling can occur. The most widely accepted theory on the settling of the sludge flocs in hindered settling conditions is the solids-flux theory. (Metcalf and Eddy, 1991, p. 221). It has been shown that the content of small solids in the secondary clarifier depends on the sludge concentration in the aeration tank, because at higher sludge concentrations smaller particles have a better chance to be adsorbed by macroflocs (Otterpohl and Freund, 1992). Sludge characteristics also exhibit pronounced seasonal variations.

3.3.3 Factors affecting secondary clarification, sludge characteristics and production

Akça et al. (1993) stated that the settling properties in secondary clarifiers depend on the type of dominant microorganisms in the biomass and the amount and characteristics of the extracellular enzymes produced by microorganisms. These factors are closely related to the operational conditions of the aeration basin, such as sludge age. In general, sludge rich in PAOs settle better because the bacteria form a dense cluster and intracellular polyphosphate, which increases the sludge density (Cuevas-Rodrigues et al., 1998; Schuler et al., 2001). Processes with combined chemical phosphorus precipitation may have even better settling characteristics (Martins et al., 2004). It has also been suggested that enhanced biological phosphorus removal processes are less susceptible for filamentous bacteria because of alternating anaerobic-aerobic conditions. Jönsson et al. (1996) reported better settling characteristics and a lower filament index in the EBPR sludge compared with simultaneous precipitation. Kappeler and Gujer (1994) suggested that extended anoxic sludge blankets in the secondary clarifiers could hinder the obligate aerobic filamentous micro-organisms in their competition against facultative aerobic floc-formers and might therefore improve the settling of the sludge. On the other hand, Lee et al. (1996) did not observe any improvement in the settling characteristics between a BNR plant and a conventional activated sludge plant, but they observed clearly less variation in the DSVI in the BNR plant. Yun et al. (2000) also found a correlation between the degree of denitrification and sludge settleability. On the other hand, Eikelboom et al. (1998) stated based on a study in more than 200 European WWTPs that nutrient removal activated sludge plants have more bulking problems than conventional activated sludge plants. In EBPR plants, the higher weight of the sludge flocs compensates the effect of the larger number of filaments.

Also sludge production is affected by several factors. Barker and Dold (1996) mentioned operational parameters such as sludge age and loading pattern, wastewater characteristics (e.g. strength and unbiodegradable particulate fraction), system configuration, chemical addition and effluent quality. They also observed that anaerobic-aerobic (and to a lesser extent anoxic-aerobic systems) produce less sludge than totally aerobic systems due to COD "loss" during hydrolysis under anaerobic or anoxic conditions, RBCOD fermentation, and VFA sequestration by PAO s. Lee et al. (1996) observed a 17% lower sludge yield in a BNR process compared with a conventional activated sludge process. On the other hand, Jardin and Pöpel (1996) observed that enhanced biological phosphorus removal resulted in an additional dry solids production of 3 g SS/g P, but they did not find any indications for a decrease in organic sludge production in the anaerobic-aerobic system compared with only aerobic. Baetens et al. (1999) stated that sludge production and sludge age are temperature-dependent, because at lower temperatures, less substrate is needed for maintenance, leaving more substrate available for growth. Moreover, decay processes are slackened. SRT of the process is naturally only decreased at lower temperatures, if MLSS is maintained constant by wasting more sludge.

3.3.4 Effects of equalization and prefermentation on sludge characteristics

Very little literature could be found on the effects of equalization or prefermentation on secondary clarification and sludge production. According to Bolmstedt and Olsson (2005), the use of an equalization basin can lower the effluent load of suspended solids. Eikelboom et al. (1998) stated that during hydrolysis, lipids and fats form higher fatty acids that filamentous bacteria *M. parvicella* uses for growth. Thus, prefermentation could result in an increased availability of substrate for *M.parvicella*. Nevertheless, hydrolysis of lipids and fats will also take place in anaerobic parts of primary clarification and in the anaerobic zone. On the other hand, VFA produced in the prefermentation could enhance EBPR, which in turn would improve the settling characteristics of the sludge.

Other studies have also been focusing on the factors affecting the growth of filamentous bacteria that in turn would affect the settling properties. Martins et al. (2004) wrote that dynamic feeding patterns, i.e. flow and load variations, can favour the growth of filamentous bacteria because low nutrient concentrations are occurring. Moreover, some filamentous bacteria, like *M. parvicella*, can exhibit the ability to store substrate under high substrate conditions. This ability was until recently believed to give some advantage for floc-forming bacteria, but recent studies show that filamentous bacteria can have an even higher storage capacity than the floc-forming bacteria (Beccari et al., 1998; Martins et al., 2003). Krishna and Van Loosdrecht (1999) interpreted the storage capacity from a population dynamics point of view by stating that in systems subjected to dynamic feeding, organisms that do not accumulate storage polymers cannot maintain cell processes and will decay during the famine period. When the substrate feeding is dynamic, the substrate uptake rate and balanced growth are more important than fast growth.

McCue et al. (2006) observed that when using prefermentation instead of primary clarification, the sludge production increased, but the WAS yield was lower with prefermentation. The authors attributed this to the production of forms of COD compounds with lower yield characteristics, such as acetic acid and propionic acid during anaerobic fermentation metabolism. Van Loosdrecht et al. (1997) observed that storage polymer formation leads to a higher sludge production, provided the polymer is not oxidized. On the other hand, growth on the storage polymer results in lower sludge yield than direct growth on the soluble COD.

3.3.5 Findings from the Pihlajaniemi WWTP

The results concerning sludge characteristics and production are presented in Appendix IV. It was observed that sludge settleability was better in the equalization/prefermentation process train. Moreover, the settling properties improved when more inert material was fed to the process. A significant negative correlation in both process trains was also found between temperature and DSVI. This might be a consequence of the improved enhanced biological phosphorus removal, as suggested by Schuler et al. (2001), but in the primary clarification process train, no difference in EBPR was observed between summer and winter. So it is more likely that the sludge settling properties improved because the degradation process of the organic matter was more complete in warm temperatures. The bacterial population might have varied with temperature as well. In addition, a significant positive correlation between sludge age and sludge settling properties was found in the equalization/prefermentation train, supporting the above-mentioned hypothesis (Akça et al., 1993), although the sludge age was kept quite constant for just over 20 days by altering the MLSS. The analysis of sludge with a microscope did not reveal any significant changes in the presence of filamentous bacteria. Overall, there were few filamentous bacteria in the sludge.

The waste activated sludge production and yield were lower in the equalization/prefermentation process train than in the reference process train. The yield in the equalization/prefermentation process train was 0.42 kgSS/kgCOD, and in the reference process train it was 0.59 kgSS/kgCOD, thus supporting the findings of Barker and Dold (1996) and McCue et al. (2006). The sludge yield calculated per kg of SS in the feed water was 20% higher in the primary clarification process train. The yield was presumably also affected by the small difference in sludge age among the process trains. The author's view is that the constant process conditions obtained by the flow equalization also contributed to the lower sludge yield by enhancing the biological degradation process. The effect of temperature on sludge production (Baetens et al., 1999) could not be fully affirmed, as can be seen in Figure 3.10, where the monthly removed waste activated sludge amount (kg) corrected with the change in the amount of sludge in the process is presented for both process trains with the wastewater temperature. In March 2003, sludge was escaping from the secondary clarifier in the primary clarification process train and WAS flow rate was increased by 50 % during the whole month. In August 2003, the WAS flow rate in the equalization/prefermentation process train was dropped rapidly, because the process was practically running out of sludge. The WAS flow rate was kept at low level during four months in order to increase the amount of sludge in the process. During the rest of the study months the volumetric flow rate of WAS was the same in both process trains. The small differences in the figure are due to different MLSS between the lines and changes in the amount of sludge stored in the process.

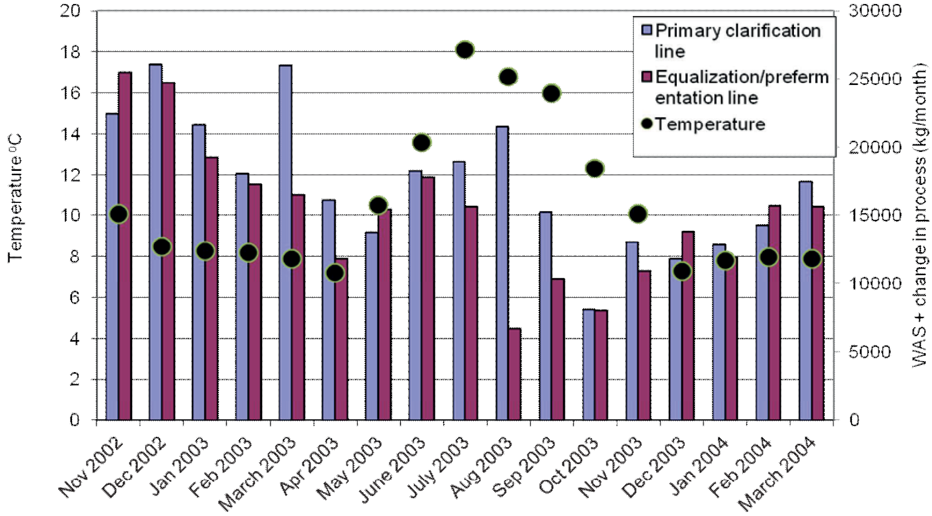

Figure 3.10 Removed waste activated sludge corrected with the change in quantity of sludge in the process monthly with the temperature at the end of each month.

3.4 The effects of flow equalization and prefermentation on energy consumption

3.4.1 Introduction

A growing concern has been given to energy consumption in different parts of the process in the wastewater treatment. In general, aeration is the part of the treatment process that consumes the biggest part of the total energy need. Altogether, biological treatment consumes 50–80% of the electricity used for wastewater treatment. This percentage contains aeration, mixing and pumping for different sludge recycles. After aeration, the biggest energy user is sludge treatment, which consumes approximately 20% of all the electricity consumed at the wastewater treatment plant. Sludge treatment can also use a major part of the heat produced at the plant (Kjellén and Andersson, 2002). An attempt to assess the overall impact of equalization and prefermentation on energy need is presented in the next section.

3.4.2 Energy consumption

The energy efficiency of aeration is greatly affected by the type of aerator, basin depth, aerator and mixer configuration, and dimensioning of pumps and compressors. Equalization can potentially improve the efficiency of the aeration system by limiting the need for air flow adjustments. Also, the dimensioning of pumps and compressors is easier and a lower peak factor for the dimensioning of pumps and compressors can be used. The pumps are operated within the optimal area and thus energy consumption is decreased. The highly fluctuating influent flow leads to over-dimensioning of all of the processes at the plant. Thöle et al. (2008) pointed out that the power input for the agitators to prevent sludge deposition is a function of the settling characteristics of the sludge. In addition, they also showed that significant savings in the air supply could be achieved by operating the plant with temperature-adjusted sludge age. Plants with a long sludge age (which is required for nitrification) consume more energy than plants with a short sludge age. This is due to higher oxygen requirements and power consumption of the sludge recycling. Furthermore, in a hydraulic flow there are always losses caused by friction. Flow velocity strongly affects the losses, which, in the case of pumped wastewater, must be compensated for by increasing the power of the pumps (Kjellén and Andersson, 2002). If the flow velocity can be kept more constant, the highest losses will be eliminated as well.

In sludge treatment, energy is used for transport, mixing, thickening and dewatering the sludge. In order to reduce the energy consumption for sludge treatment, the general goal is to decrease the amount of sludge. The dry solids content of the sludge should also be as high as possible, nevertheless taking into account the limitations of pumps and different treatment processes. Gravitation thickening has a low energy demand, but all the other methods of thickening and dewatering of sludge require a considerable amount of energy, from 15–60 kWh/t TS. (Kjellén and Andersson, 2002). In wastewater treatment plants where sludge digesters are used, part of the energy and heat need can be compensated with biogas and heat produced in the digestion. Biogas production of a sludge originating from a long sludge age plant is fairly low (Kjellén and Andersson, 2002), whereas primary sludge has a better biogas yield. Moreover, the biogas production of primary sludge is affected by the retention time of the primary clarifiers. Kjellén and Andersson (2002) showed that a longer retention time increased the amount of primary sludge produced. When the retention time was increased from 0.5 h to 2 h, the primary sludge production was increased from 30–38 g/PE/d to 45–50 g/PE/d. Andreasen et al. (1997) observed a 25% decrease in gas production from anaerobic digesters due to the introduction of primary sludge hydrolysis. On the other hand, Jönsson et al. (2008) observed in their experiments that no significant change occurred in the methane production potential after hydrolysis of primary sludge.

3.4.3 Energy balance from the Pihlajaniemi WWTP

In our case, the energy needed for the equalization pump was very small, only about 0.4% of the total energy demand of the whole plant. This was partly due to the fact that the pump was stopped whenever the influent flow rate increased to the level where the overflow weir could be used. Also, it should be remembered that only half of the influent flow was equalized. Nevertheless, the energy consumption of the equalization basin could be even further decreased by the use of a primary treatment bypass (Dold et al., 1984).

The most significant improvement for energy consumption was achieved in the sludge treatment. The amount of waste activated sludge was reduced by 8% and the primary sludge by 25% during the second 9 months of the study. If we assume that the energy need of sludge treatment was between 10% and 20% of the total energy consumption at the plant, the reduction would mean 1-3% of the total energy consumption of the plant. Moreover, contrary to expectations, the sludge dewatering was easier – during the study, the dry solids content after the centrifuge varied from 21-30% (averaging 23%), whereas during the years 2001–2002 (before the study), the values were 18–23% (averaging 21%).

The aeration efficiency also improved, assumingly due to more constant conditions in the aeration basin, but this did not reduce the energy consumption, because at the same time, the load to the equalization/prefermentation process train was increased. Moreover, at the steady plant influent flow rate, hydraulic losses were decreased. Nevertheless, this was not very significant at the Pihlajaniemi WWTP because equalization was not removing the highest peaks of flow, only the diurnal variation. Furthermore, the flow through the plant is mainly gravitational. It should also be stressed that all the improvements were not fully materialized at the Pihlajaniemi plant when introducing equalization, because the equipment, e.g. pumps and compressors, were not changed.

The Pihlajaniemi plant is not using digestion for sludge treatment, and for this reason, the production of biogas was not an issue. Nevertheless, it can be assumed that biogas production of the sludge from the equalization/prefermentation process train would be lower than that of the sludge from the primary clarification process train, due to the smaller amount of sludge, the smaller proportion of raw sludge, and the higher degradation level of the organic matter in the WAS. It seems, however, that the influence of primary sludge hydrolysis on the digester gas production is not yet fully understood (Andreasen et al.1997; Jönsson et al., 2008).

3.5 Prerequisites for flow equalization and prefermentation

3.5.1 Introduction

The influent wastewater characteristics and the flow and load pattern determine the need for pre-treatment at the WWTP. In case of the BNR process, even more attention has to be paid to the feed water of the process, because bacteria capable of nutrient removal require specific conditions. Therefore, it is useful to gather information about the wastewater's origins, sewer network and the plant influent. Usually some regularity in the influent flow and load variations can be found on a diurnal, weekly and seasonal level. Diurnal variation refers to the fluctuations occurring every 24 hours. The benefits achieved by the implementation of equalization in the plant will depend on the typical flow and load pattern. Also, the magnitude of these variations is relevant when assessing the use of equalization. Furthermore, the influent wastewater characteristics and their seasonal variations give some indication of the usefulness of prefermentation.

In the following, first, the influent flow and load variation, its magnitude and different causes are highlighted. Secondly, in-sewer transformation processes and their consequences on the wastewater characteristics at the plant inlet are presented. Finally, experiences from the Savonlinna sewer network and influent wastewater characteristics in relation to the requirements of the BNR process are discussed. The beneficial characteristics of the BNR feed water were already discussed in Section 3.1.5.

3.5.2 Assessing the usefulness of flow and load equalization

The flow and load pattern at the plant inlet is the sum of flow and load components entering along the length of the sewers. The sewer system receives substrate inputs from the wastewater discharges, rain water intrusion and leakages. The diurnal variations are usually caused by the domestic or industrial wastewater discharges, whereas seasonal variations depend on the climate and the conditions in the sewer network. For the moment, no common criteria have been developed for assessing the amount of variation where equalization would be useful (cf. Section 3.1.5 for prefermentation).

3.5.2.1 *Diurnal and weekly variations*

The sum of domestic wastewater discharges usually shows two clear flow peaks during the day, whereas the late night is the time for minimum flow rate. Wastewater discharge from a single household, on the other hand, is variable to a high extent (Friedler and Butler, 1996). Therefore, if the population is very homogenous in its habits and daily rhythm, the peaks are magnified, whereas in a heterogenous population, the peaks tend to level out. Also, in growing sewer systems, the hydrographs of wastewater arriving from different parts of a city will be superimposed due to phase shift (Kracht and Gujer, 2005). In general, plants with a high industrial load have bigger weekly variations than a plant with little or no industrial load (Harremoës et al., 1993). Furthermore, flow rates during weekends are usually lower than those during the week. A thorough assessment of the diurnal flow variations is useful when the implementation of equalization is considered.

It is important to notice that the load pattern of different pollutants (e.g. organic matter, ammonium, and phosphates) may differ from the flow pattern. Pollutants are generated from different sources within a household. The hour of usage of each appliance and their proportion of the whole wastewater flow at the time determine the concentration pattern in the wastewater discharge. The appliance volume, pollutant load, and frequency of use data are highly variable (Friedler and Butler, 1996). According to Butler et al. (1995), based on data from Great Britain and Malta, the WC and the kitchen sink are the main sources of BOD load, followed by washing machines and showers, whereas ammonia is generated almost solely from WCs and phosphorus is mainly coming from washing machines and wash basins. These findings were supported by Henze (1992) who showed that the sources of nitrogen in raw municipal wastewater were primarily human excretion, of which some 75% is excreted as urea while the rest is organic nitrogen. The polyphosphates are important components of textile washing powders and other detergents. Phosphorus is also contained in many organic substances, e.g. nucleic acid and phosphoric lipids, and thus the organic phosphorus fraction also originates from organic chemicals such as pesticides and detergents (Maurer and Boller, 1999; Rössle and Pretorius, 2001).

3.5.2.2 Seasonal variations

As emphasized in Section 3.1.4, seasonal variations can rarely be flattened with an in-line equalization basin. Due to the magnitude of variation and the quality of wastewater, a separate storage tank is preferred. During a runoff event, the flow may increase by a factor of 10 or 100 (Nielsen et al., 1992). Water, air and pollutants enter the sewer system in the form of leakages or from the combined sewer systems. Rain water intrusion is linked to a specific rain event, which can be characterized by its intensity, duration and the length of the preceding dry weather period. However, Pitt et al. (2007) pointed out that typically there is no direct linear correlation between rainfall and influent wastewater flow to the treatment plant. In addition, other variables, such as temperature, wastewater pollutant concentration and solids accumulated in a sewer system, influence the quality of the wastewater arriving at a wastewater treatment plant.

In the case of combined sewers, the contribution of rain water to the total load can be significant, entailing high variability. Combined sewers are usually constructed in urbanized areas, where runoff from the streets cannot be discharged to a watershed without treatment. In a combined system, run-offs from the roofs, yards and streets are collected. According to Gromaire-Mertz et al. (1998), the contribution from runoffs was approximately 30% of the SS and COD loads, 20% of the VSS loads, and less than 20% of BOD_5 loads. Mikosz and Rybicki (2008) observed similar results, with rain water causing 5% of the BOD load, 14% of the COD load, and 36% of the SS load. The rain water entering the combined sewer system did not contain any nitrogen nor phosphorus. On the other hand, Vaze and Chiew (2004) observed that street run-offs contained phosphorus and nitrogen mainly associated with particulates.

An important part of the load can originate from sediments in the sewer system. The quality and quantity of sediment in the sewer systems are not very well known and vary significantly depending on the age, slope and flow conditions in the system. Gromaire-Mertz et al. (1998) calculated in a full-scale combined sewer study that during a wet weather period, 40–60% of the SS and COD load originated from sediments, against 20-30% from runoffs and 10-40% wastewater. In the case of BOD and VSS, the majority of load originated from sediments. Vanrolleghem et al. (1999) suggested that in sewer reaches where sediments are present, they can be considered as an unlimited resource, and during a rainfall event, they can account for a majority of the load arriving at the plant. Nevertheless, it should be noted that not all sewer reaches contain sediments. Ahyerre et al. (2001) observed an immobile layer composed of fibers and organic matter over the coarse mineral deposits in the trunk sewers.

Groundwater infiltration is either caused by leaking pipe joints, cracks in the pipe wall, or by ill-connected drainage lines (Huisman et al., 2004). Ground water infiltration causes a steady base flow to the system, which can be significantly influenced by the hydrological situation and thus varies throughout the year. The determining parameters for infiltration are the groundwater level, the surface area of the pipes wetted by groundwater, and the leakage factor. In Germany, it has been determined that the amount of infiltrated water varies between zero and several hundred percent of the sewage flow (Schulz et al., 2005). The pollutant concentration in the infiltrated groundwater can often be considered negligible (Bareš et al., 2009).

3.5.3 Assessing the usefulness of prefermentation

Before the wastewater reaches the plant, it has already been transformed in the sewer system. Many attempts for a comprehensive sewer model have been made in last decades (Gall et al., 1995; Hvitved-Jacobsen et al., 1998; Abdul-Talib et al., 2002; Hvitved-Jacobsen et al., 2002; Vollertsen et al., 2005). But due to the variability in wastewater composition and the complex transformation and transport processes in the bulk water, biofilms, sediments, and gas phase, the conditions in the sewer network and the consequences to the wastewater quality are difficult to predict (Vollertsen et al., 2005, Hvitved-Jacobsen et al., 2002). Aerobic conditions will favour hydrolysis and biological oxidation of biodegradable organic material, whereas anaerobic parts of the sewer network will be favourable for VFA production. Thus, the benefits of adding a prefermentation tank prior to the biological process should be considered with sufficient knowledge of the quality of the organic matter. It is noteworthy that the quality may vary significantly within a day or between seasons.

3.5.3.1 Aerobic or anaerobic conditions?

Hvitved-Jacobsen et al. (1998) argued that conditions in gravity sewers can remain aerobic owing to re-aeration. Usually the conditions in gravity sewers are favourable for a good oxygen transfer. On the one hand, the oxygen concentration in the wastewater is low due to the oxygen consumption of suspended and attached biomass. On the other hand, the oxygen concentration in the sewer gas phase is close to the atmospheric concentration because the slotted manhole covers and the water drag force stimulate a frequent refreshment of the gas phase (Huisman et al., 2004a). Oxygen transfer is based on eddy diffusion, which is affected by turbulence due to the flow conditions. The oxygen transfer increases, e.g. when the flow velocity, the water depth, or the surface roughness increases (Moog and Jirga, 1999; Pai et al., 2000; Balmér and Tagizadeh-Nasser, 1995; Huisman et al., 2004b). The presence of surfactants, grease or oil, on the other hand, can significantly reduce the gas-liquid transfer. Anaerobic conditions may occur in pressure sewers, in full flowing gravity sewers, or in gravity sewers with a low slope (Hvitved-Jacobsen et al., 2002). Without an oxygen supply from an air phase, the oxygen in the wastewater is usually depleted within a few minutes (Nielsen et al., 1992).

Gudjonsson et al. (2002) showed that dissolved oxygen concentrations in the sewers have a strong diurnal variation and depend significantly on wastewater composition and temperature. Based on sewer network measurements and simulation, they showed that as the wastewater temperature increased from 9 °C to 14 °C, the conditions changed from solely aerobic to alternating aerobic-anaerobic. Driven by the oxygen deficit, the re-aeration rate increases as the temperature increases, so the lower dissolved oxygen can be explained only by increased microbial activity. The dissolved oxygen concentration in the bulk water in the sewers was above 2 mg/l in the morning, but it decreased rapidly to close to 0 mg/l as the concentrated wastewater produced during the morning hours reached the sewer.

3.5.3.2 Transformation processes of pollutants in sewers

<u>Biodegradable organic matter</u>

Readily biodegradable organic matter will be produced or consumed, depending on the conditions in the sewers. In the presence of sufficient oxygen, aerobic degradation of organic matter occurs rather rapidly and with a high biomass yield. In optimum conditions, the wastewater biomass has doubling times as low as a few hours. Aerobic degradation of organic matter takes place in biofilms, sediments and the bulk water (Hvitved-Jacobsen et al., 2002; Vollertsen et al., 2005). There are different views of how much of the organic matter can be converted to smaller molecules or consumed in the sewer system. A concern has even been raised that in a large catchment, all the readily biodegradable material could, as a result of aerobic sewer reactions, be consumed and biological nutrient removal at the plant could be at risk (Raunkjer et al., 1995).

Full-scale studies and theoretical calculations have shown that in sewer systems, the removal of dissolved organic carbon (DOC) varies between 14% and 50% (Gall et al., 1995; Huisman et al., 2004a; Chen et al., 2001). Raunkjer et al. (1995) observed DOC removal rates of 9 to 15 mg/l/h. They also determined that primary compounds removed were dissolved carbohydrates and acetate. On the other hand, Flamink et al. (2005) calculated that in the Netherlands, with a typical hydraulic retention time in the sewer system of 2 hours, the DOC removal rate would be an order of magnitude lower than the diurnal fluctuations in COD concentrations, which in this case was 380640 mg/l. Siegrist and Tschui (1992) observed at a Swiss treatment plant that the COD load in summer was 15% lower than in winter. The authors attributed this partly to increased COD degradation in the sewer system at higher temperatures. Ahnert et al. (2005) conducted a study where the overall degradation in a sewer system was estimated with long-term (almost seven years) plant influent data. They assumed that at temperatures lower than 4 °C, no microbial COD degradation occurred in the sewer systems. Since total phosphorus is not transformed or degraded in the sewer system, they used the ratio COD/P to evaluate the amount of COD degradation in the sewers. The authors showed that COD degradation was strongly correlated with temperature. The COD/P_{tot} ratio reduced about 20% from 8°C to 19°C, meaning that 20% of the COD was removed in the sewer system at the warmest temperatures. The authors derived two different plausibility checks with a surface aeration rate (k_La) and oxygen uptake rate (OUR) for the calculated degradation. The values were in the same range as the values from other studies. Tanaka and Hvitved-Jacobsen (1998) showed in laboratory scale tests that the aerobic, heterotrophic biomass activity was maintained in the water phase during the six-hour phase in anaerobic conditions, suggesting that the aerobic activity would not suffer from alternating conditions described by, e.g. Gudjonsson et al. (2002).

Under anaerobic conditions, fermentation may occur, i.e. organic matter acts as both donor and acceptor of electrons (Vollertsen et al., 2005). In this case, anaerobic hydrolysis follows the phases described in Section 3.1.3. Tanaka and Hvitved-Jacobsen (1998) observed in laboratory scale and in a pressure sewer, the net production of a readily biodegradable substrate of 1.5–1.8 g $COD/m^3/h$. Anaerobic transformation processes in the sewers may cause the RBCOD and VFA concentrations to fluctuate considerably more at the plant inlet than the COD or BOD concentration. Armiger et al. (1993) observed that VFA concentration in the primary effluent changed by nearly 300% over a typical diurnal cycle. This was the result of the increased production of VFA in the sewer system due to a lower night time flow and the improved performance of prefermenter due to an increased retention time at the same time.

Ammonium

In order to initiate nitrification in the sewer system, a very long retention time would be needed. Gall et al. (1995) observed that in aerobic conditions, NH_4^+-N concentration decreased in raw wastewater after two days. On the other hand, the retention time inside biofilms and sediments can be longer than the hydraulic retention time of the system. The permanent presence of oxygen is needed to increase the possibility of some nitrification in the biofilm. Moreover, the organic loadings should be low (Nielsen et al., 1992). Ahnert et al. (2005) showed that in a sewer system of the city of Dresden (520,000 inhabitants), TKN load decreased by 15% during a warm period. According to the authors, the drop could be explained only by nitrification followed by denitrification. The decrease in ammonium correlated well with the temperature. Contradictory results were obtained in the study of Siegrist and Tschui (1992), where a lower COD load to the plant in summertime was observed, but the ammonium load to the plant was 3% higher during summertime. The sewer network in this case was not described, but the plant was relatively small (PE 110 000). A possible explanation could be that ammonium was produced from fermentation deep in the biofilm. Moreover, hydrolysis of proteins also produces ammonium, and although a part is reassimilated for growth, a net production may occur (Nielsen et al., 1992).

Nitrates

Anoxic conditions and thus denitrification have been until recently considered very rare, unless nitrate is added to the sewer network (Abdul-Talib et al., 2002). Nevertheless, Huisman et al. (2004a) observed that even under aerobic conditions, nitrates originating from drinking water and groundwater infiltration were denitrified in the deeper parts of the biofilm. Also, Æsoy et al. (1997) pointed out that denitrifying bacteria grow well in biofilms because they are able to produce relatively large amounts of polymers. Abdul-Talib et al. (2002) showed with full-scale experiments that denitrification also takes place in bulk wastewater with accumulation of nitrite in the water phase. Ahnert et al. (2005) studied the in-sewer transformation processes with an analysis of long-term plant influent data. They demonstrated a correlation of influent wastewater nitrate concentration with a temperature ranging from 1.75 mg NO_3-N/l at 8 °C to 0.3 mg NO_3-N/l at 18 °C. The data was from the city of Dresden with approximately 520,000 inhabitants, where sewers in outer catchments are intensively re-aerated due to a high slope. The trunk sewers before arriving at the WWTP are 8 to 12 km long with a low slope. The authors concluded that sewer network design in this case offered good conditions for first nitrification, and when nitrates were present also denitrification either in the biofilms or in the trunk sewers where conditions most probably became anoxic and anaerobic.

Suspended solids

Gromaire-Mertz et al. (1998) showed that in combined sewers, the particulate fraction of the wastewater increased from entry to the outlet in the sewer system. The authors explained the increase by erosion of sewer sediments and biofilm and adsorption of pollutants on particles. Particulate matter can also increase in the sewers as a consequence of biomass decay (Løkkegaard-Bjerre et al., 1995).

Phosphorus

Total phosphorus should not decrease or increase in a sewer network, since no transformation processes are known for total phosphorus (Ahnert et al., 2005). Nevertheless, the amount of ortho-phosphates may change, since polyphosphate and organic phosphates can revert through a slow-rate hydrolysis process to soluble ortho-phosphates (Maurer and Boller, 1999; Rössle and Pretorius, 2001). Moreover, some phosphates will be taken up by the biomass growth. Particulate phosphorus may also be absorbed by the biofilms and also be detached (Æsoy et al., 1997).

3.5.4 Plant influent wastewater and in-sewer transformation processes at the Pihlajaniemi WWTP

3.5.4.1 Characterization of the influent wastewater at the Pihlajaniemi WWTP

The quality and quantity of the plant influent was monitored during 18 months as described in Section 2.9. The RBCOD and VFA concentrations in the influent water were low. The measured concentrations and their variation are described in more detail in Appendix V. There was evidence that significant amounts of VFA and RBCOD were produced in the sewer network in summertime and also in wintertime, when the flow rate was normal for dry weather. In addition, some evidence was found that the VFA potential of the plant influent wastewater was better during the winter. Nevertheless, at high flow rates, very little or no VFA production was detected, assumingly due to the shortened anaerobic retention time in the sewer network (V). According to the limits stated in the literature (Pitman, 1990; Abu-ghararah and Randall, 1991), the readily biodegradable content in the influent wastewater was insufficient for biological nutrient removal. Moreover, the RBCOD concentration in the plant influent varied between 1% and 6% of the total COD, which is lower than was observed in other studies in Switzerland and Denmark (Henze, 1992; Sollfrank and Gujer, 1991; Vollertsen, 2006). This was probably due to biological oxidation taking place in the sewers.

The COD:TKN and COD:TP ratios in the influent wastewater are shown in Table 3.4. In comparison with the values given in the literature (Pitman, 1990; Rössle and Pretorius, 2001; Cuevas-Rodrigues et al., 1998), it can be observed that the carbon-to-nutrient ratios are suitable for biological nutrient removal, although the total COD values should be handled with some scepticism. In conclusion, the plant influent could be classified as suitable for BNR after a pretreatment where more RBCOD would be produced from SBCOD through hydrolysis..

Table 3.4 Carbon-to-nutrient ratios in the influent wastewater during the study at the Pihlajaniemi WWTP.

n=43	Average	95% percentile	5% percentile
COD_{Cr}:TKN	12	14.7	9.9
COD_{Cr}:TP	61	73.6	49.1
COD_{Cr}:NH_4-N	16	22.4	11.8

3.5.4.2 Seasonal, daily and diurnal variations of Pihlajaniemi plant influent

The seasonal variation of the plant influent flow is presented in Appendix V. The expansion of the sewer network and the control of the storage tank and the recycle flows have resulted in a lower diurnal variation in the influent flow. To illustrate this, the ratio between minimum and maximum flow rate during dry weather period was calculated. The maximum hourly influent flow rate in February 1995 was 31.2 times the minimum hourly flow rate. In February 2004 the ratio was only 4.5. Wastewater from the Punkaharju storage tank is usually pumped to the plant during the night in order to even the influent flow. A clear evidence of the usefulness of this operation could be seen during the study when the pumps in the storage tank were out of order during two months and the tank was temporarily emptied during the day. Figure 3.11 presents the influent flow rate at the plant during one week in January, when the storage tank was emptied during the daytime, and during one week in March, when storage tank was again operated normally. When the storage tank was emptied during the day, the ratio between the maximum and minimum flow rate increased from 4.5 to 13.6.

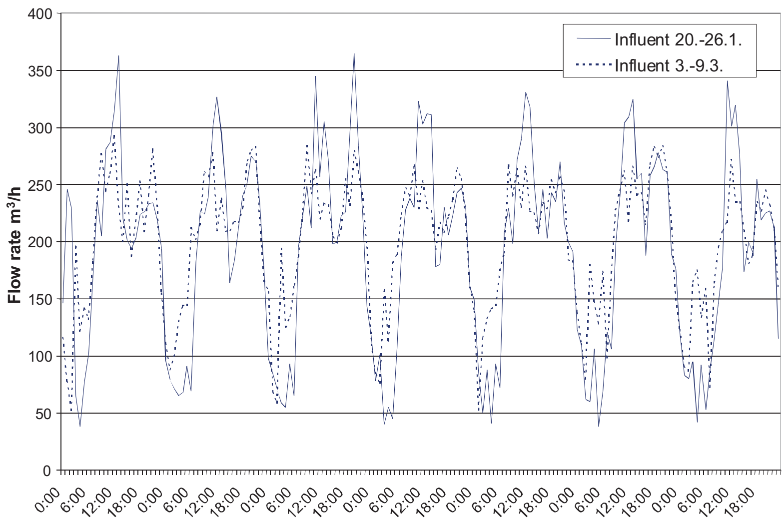

Figure 3.11 Weekly influent flow rate 20.-26.1.2003 and 03.-09.3.2003. In January, the storage tank was emptied during the day and in March during the night.

The influent load is shown in figures 3.12, 3.13, and 3.14. The load during the study and the monthly average values from the three years prior to the study are plotted. It can be seen that organic matter, ammonium, total nitrogen, and suspended solids show some seasonal variation with the highest loads in March and April corresponding to the snow melting period. During this period, it can be assumed that no biological transformations were taking place in the network due to the low temperature and high flow rates. In comparison with other parameters, total phosphorus is very stable, especially the long term average values, as also observed by Ahnert et al. (2005). Nevertheless, a slight increase in the phosphorus load is visible during snow melting periods as well as several peaks in the data from the study period. These increases can be attributed to the scour of biofilms and sediments from the sewers during wet weather periods. Furthermore, it should be noted that nitrogen and COD loads are slightly lower during summertime, whereas the SS load is lower during wintertime and higher during summertime. The first was presumably due to the biological activity in the sewers in warm temperatures and with sufficient retention time, as discussed in Appendix I. The latter, on the other hand, could be due to the fact that rain water intrusion was more important during the warm period, conducting a high load of SS into the sewer (Gromaire-Mertz et al., 1998; Mikosz and Rybicki, 2008).

Moreover, the nutrient load contained in the reject water was relatively high during the whole study, which also influenced the plant influent or the wastewater entering the pre-treatment. The reject from sludge thickeners contained a nitrogen load corresponding to 4-15% of the influent load and 1-3% of the phosphorus load, for centrifuge the values were 1–2% and approximately 0.5%, respectively. The nutrient load in the filter wash water varied greatly, being 1–5% of the plant influent load for nitrogen and 1–26% for phosphorus.

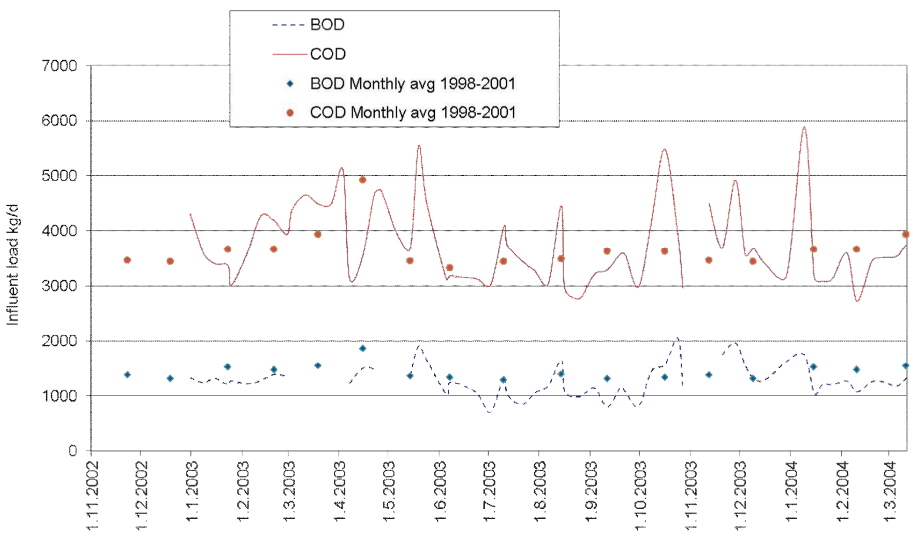

Figure 3.12 BOD and COD plant influent load during the study and the monthly average loads from 1998 to 2001.

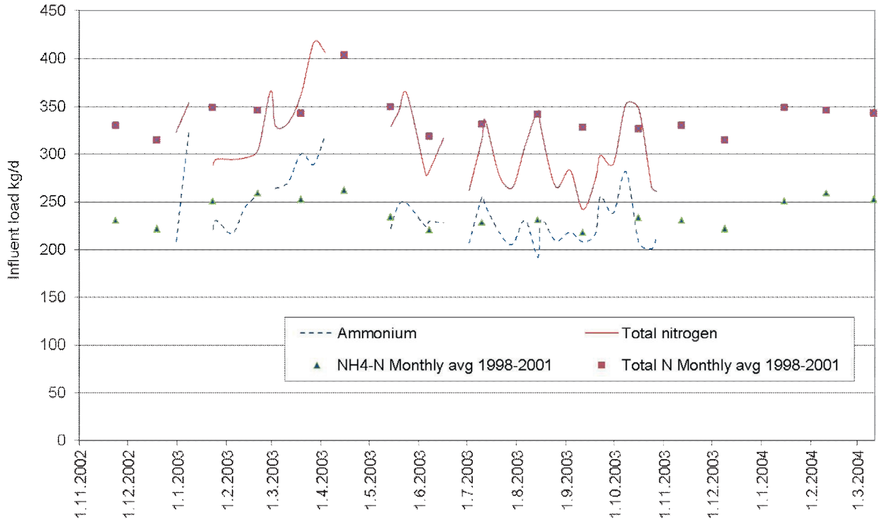

Figure 3.13 Ammonium and total nitrogen plant influent load during the study and the monthly average loads from 1998 to 2001.

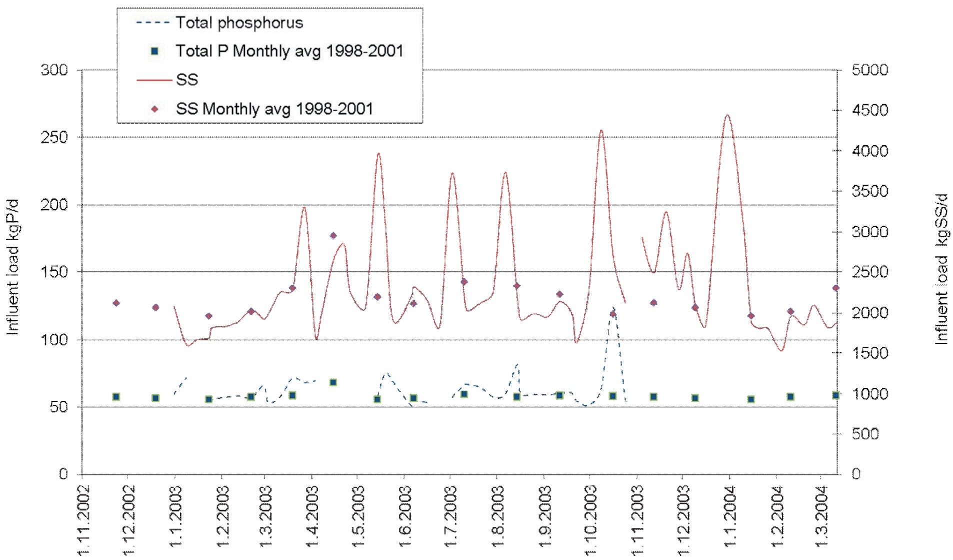

Figure 3.14 Total phosphorus and SS plant influent load during the study and the monthly average loads from 1998 to 2001.

The daily variations in the influent load were observed between working days and weekends. The load was lower during weekends, as shown in Table 3.5. It can also be assumed that the smaller the difference between weekdays and weekends, the bigger the portion of the pollutant originating from other sources than municipal wastewater (e.g. street run-offs into combined sewers). This would mean that especially COD and SS are also coming, e.g. from street run-offs, as also proposed by Gromaire-Mertz et al. (1998) and Mikosz and Rybicki (2008).

Table 3.5 Average weekend loads (the whole study) and the percentage of the working day loads (the whole study).

	Weekend load	Percentage of the working day load
	kg/d	%
COD_{Cr}	3720	99
BOD_7(ATU)	1139	86
NH_4-N	229	94
Total nitrogen	293	92
Total phosphorus	56	88
Suspended solids	2205	95

Diurnal variations of concentration and load in the plant influent, i.e. variations within the day, were higher for soluble COD and phosphate phosphorus than for total COD, ammonium and suspended solids. The same observation was made by Harremoës et al. (1993), who also pointed out that carbon-to-nutrient ratios will consequently change during the day. Figure 3.15 presents an example of diurnal change of the carbon-to-nutrient ratio during the study. Our measurements show that in the case of soluble COD and phosphorus, the peak for diurnal concentration and flow occurred at the same time, amplifying the variations in load. The phosphorus concentration and load peaked around 6 p.m. and then decreased slowly, suggesting that washing machines were mainly used in the afternoon and in the evening. The ammonium concentration was highest during the day from 9 a.m. to 2 p.m. This can be explained by the fact that during the daytime, other appliances were used less and wastewater from WCs accounted for a bigger proportion of the total discharge. The ammonium load, on the other hand, was high during the whole day from 9.a.m. to 9.p.m., corresponding to the period when a WC is mostly used.

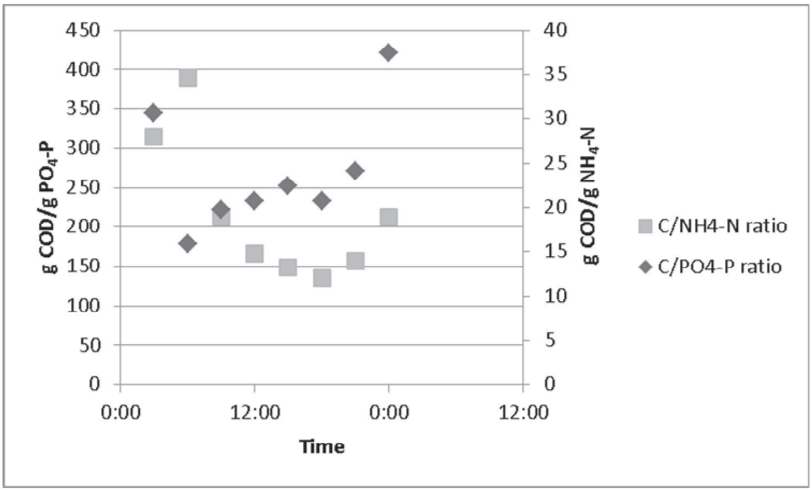

Figure 3.15 Variation of the carbon-to-nutrient ratio in the influent wastewater during one day.

3.5.4.3 In-sewer transformation processes in the Savonlinna sewer system

The approach in the study in Savonlinna did not include any measurements in the sewer network. Assumptions about the sewer conditions were made based on the influent wastewater parameters. It was shown that a significant amount of VFA was produced in the sewer network when the flow rate was normal for dry weather. On the other hand, in-sewer processes were responsible of decreasing the load to the treatment plant. With our plant influent data, it could be concluded that during the warm water period, biological oxidation of organic matter took place in the aerobic part of the sewer network. Moreover, the results also suggest that nitrification took place in the sewer system. This finding was contradictory to the skeptical opinion among many scientists and operators towards significant nitrification in the sewers. Assumingly, denitrification also occurred in the anoxic parts. As in the study conducted by Ahnert et al. (2005), we also found a significant correlation between the COD_{Cr}:TP ratio and temperature (Pearson correlation coefficient R=0.52; 0.1%), suggesting that organic matter was consumed during the warm period. A significant correlation was also found in the case of total nitrogen with the TN/TP ratio decreasing with increasing temperature (Pearson correlation coefficient R=0.50; 0.1%). This suggests that nitrification and denitrification were occurring in the sewers during warm temperatures. It must be noted that the sewer network is wide-spread and the retention time can be long, as explained in Appendix V. Nevertheless, the correlation between nitrates in the influent and temperature observed by Ahnert et al. (2005) could not be confirmed at Pihlajaniemi, where NO_2-N and NO_3-N in the influent were measured only occasionally. The total phosphorus load to the wastewater plant in Pihlajaniemi was very stable throughout the study (Figure 3.14). On the other hand, it was observed the TN/TP ratio decreased also at high flow rates. This was explained by the fact that the average flow rate was higher during the warm period. Surprisingly, COD/TP did not correlate significantly with the flow rate. This observation supports the hypothesis that large amounts of organic matter entered the sewer system with rain water. The findings on in-sewer transformation processes in Savonlinna are described in more detail in Appendix V.

4 Discussion

4.1 Costs vs. benefits for equalization basin and prefermentation

In the case of the studied equalization configuration at the Pihlajaniemi WWTP, the costs of equalization were low. Because the existing basin volume was used, the only investment cost was the equalization pump. Many Finnish treatment plants have primary clarification basins. According to the study by Kangas (2004), 60% of the WWTPs with PE above 10,000 have primary clarification, and thus large number of plants would potentially have basin volume to be used for equalization. The operation cost was also small, only 0.4% of the total energy cost at the treatment plant, i.e. approximately 0.0002 €/m^3 of wastewater.

The investment cost for prefermentation was even lower than the cost for equalization. An existing pump was used for the sludge recycle in the equalization/prefermentation basin. The operation cost was approximately half of the cost of the equalization. On the other hand, the performance of the prefermentation in this study was not very good. As a matter of fact, the same amount of soluble BOD and VFA was produced in the equalization/prefermentation basin with and without a sludge recycle.

The benefits of equalization and prefermentation were already discussed in previous chapters from the process engineering point of view, so here the focus will be on the monetary value of the achieved benefits. The biggest reduction in costs was achieved in sludge treatment. The 5–15% reduction in sludge production led to 10,000–20,000 € yearly savings in sludge treatment when the costs of lime and polymer addition and composting were taken into account. The aeration efficiency was improved, but it is difficult to estimate how much more air would have been needed to reach the same effluent concentrations in an unmodified process train. Better process control and increased capacity also reduced the need for using ferrous sulphate and polymer in the secondary clarification. It is also difficult to estimate the real savings from this, especially because the plant has no nitrogen limit for the moment. Anyway, the economical balance for equalization is positive with the benefits from sludge treatment alone. With energy intensive processes such as membrane bioreactors, UV disinfection, and DAF, reductions in energy consumption achieved by optimal dimensioning would also be significant. Enhanced prefermentation in the studied application did not bring very many benefits, but it most certainly could be improved without increasing the costs.

4.2 Feasibility of prefermentation in Finnish conditions

This study demonstrated that APT application for prefermentation can be carried out very well in Finnish conditions. Moreover, prefermentation is widely used, e.g. in Canada and in Denmark, where conditions are quite similar and where several studies have proven the feasibility of prefermentation in cold climates (Vollertsen et al., 2006; Yuan et al., 2009). On the other hand, the benefits of prefermentation are partly linked to enhanced biological phosphorus removal, which is only practiced in a couple of Finnish treatment plants. The other possibility to benefit from prefermentation is to have pre-denitrification in the process train, but recently many treatment plants in Finland have chosen post-denitrification in order to improve the total nitrogen removal. The use of primary sludge hydrolysate for post-denitrification is limited by the phosphate content of the hydrolysate. Thus, at the moment, the most common process configurations and also the design traditions do not support prefermentation.

If wastewater's internal carbon is not used, some external carbon source must be added to the biological process in order to ensure sufficient input of organic matter for bioP bacteria and denitrification. Several alternatives exist, such as methanol, ethanol, acetate, sugar, glycerin, glycerol, food industry wastewater, *etc*. (Tam et al., 1992; Canziani et al., 1995; Swinarski et al., 2009; Bilyk et al., 2009). Methanol is currently the most widely used product (Bilyk et al., 2009), although the maximum specific growth rates of organisms utilizing, e.g., acetate, ethanol or sugar are significantly higher than the rate of methanol-utilizers, especially in low temperatures. This means that a longer SRT must be provided when methanol is used (Dold et al., 2007; Mokhayeri et al., 2008). Methanol also has the disadvantage of being a flammable and hazardous liquid that requires significant handling and storage facilities (Johnson and Drainville, 2009). External carbon addition is often considered as a more reliable and more stable process in comparison with prefermentation. On the other hand, the cost and the environmental footprint for methanol are significant. Hunt et al. (2009) conducted a cost and carbon footprint comparison for different wastewater treatment processes and observed that a process using ferric sulphate dosing and external carbon had the highest costs and the largest carbon footprint, whereas a step-feed BNR process ranked first for economical and environmental impact. The use of methanol and ferric sulphate contributed up to 38% of the operation greenhouse gas (GHG) emissions of the plant, with methanol usage having a larger carbon footprint than ferric sulphate usage. Moreover, Dold et al. (2007) showed that both methanol and ethanol have a strong temperature dependency with an Arrhenius coefficient of 1.13. They also observed that many denitrifying organisms are substrate specific, i.e. they can only use one or a couple of carbon sources. These aspects, together with an economical cost, may in the future help to gain interest in the use of wastewater's own carbon content and prefermentation in Finland as well.

4.3 Feasibility of flow equalization in Finnish conditions

Equalization could easily be implemented and could have a beneficial effect on the treatment performance in many Finnish treatment plants. As already mentioned earlier, a majority of big and middle-sized treatment plants have primary clarification, and the basin volume could be used for equalizing the diurnal flow variations. This would not prevent the use of the basins for primary sludge removal. On the other hand, it was shown that the expansion of the sewer network in Savonlinna has resulted in a smaller variation in influent flow. Thus, it can be estimated that the implementation of an equalization basin in a WWTP with a smaller sewer network would have more effect than was observed in the case of Savonlinna. On a more general level, the present development with growing agglomerations as well as the closing of smaller treatment plants and concentrating the treatment in central treatment units will help to even out the plant influent fluctuations (Harremoës et al., 1993). Moreover, in many cases development towards centralized systems will leave old basin structures that can be used as equalization basins. Recently, more attention has been paid to sewer network renovation in order to decrease the amount of leakages. In spite of continuous investments in the sewer network, it is difficult to decrease the amount of rain water intrusion as the network is ageing. Overall, it can be estimated that the need for equalization will still exist in the future.

In addition to an actual equalization basin, there are many ways of contributing to the equalization of the flow and load pattern entering the biological process. The whole wastewater treatment system – including the wastewater treatment plant, the sewer network and even connected households – should be studied in order to recognize the beneficial actions. At the treatment plant, a great deal of attention should be paid to the reject water flows and the time when they are introduced to the plant inlet. The point in the process where the reject flows are added should also be carefully chosen, e.g. reject water from a centrifuge usually contains a lot of VFAs, except for digested sludge. VFA-rich reject water could be introduced directly in the beginning of the biological process instead of at the plant inlet. In smaller plants, where sludge dewatering is only used for a short period of time during the day, the control of the reject water recycle is very important. Moreover, the use of different bypasses in order to equalize the flow and load should be considered. In addition, optional strategies to deal with storm flow events have been proposed, such as modification of the wastewater feed pattern to the BNR aeration tanks, adjustment of the operating mode of the biological process, manipulation of the process biomass storage, and diversion of the wastewater flow around the activated sludge process with or without treatment (Pitt et al., 2007). These possibilities should be carefully considered.

The sewer network volume will naturally level out part of the flow variations, but in many cases the sewer volume could be used in a more efficient way. Large interceptor sewers can be effectively used as storage tanks to dampen peak diurnal dry-weather variations, but a nightly or weekly drawdown of the interceptor system is necessary to flush sludge that has accumulated during the storage period. In some larger cities, such as Chicago, wastewater tunnels are used to store wet-weather flow and maintain a nearly constant flow to the treatment plants (Vesilind, 2003). The volume existing at the pumping stations can also be optimally used, and actual equalization basins can be built in the network.

4.4 Reliability and validity of the study

Although the study was carried out only in one wastewater treatment plant, it can be estimated that the results can be generalized for other plants as well. The wastewater arriving at the Pihlajaniemi WWTP is typical wastewater from municipal sources. The large sewer network in Savonlinna flattens the flow variations and in some cases acts as a prefermenter. Therefore, it can be estimated that in a smaller sewer network, the benefits of equalization and prefermentation would be more pronounced.

Measurements are always prone to error. Systematic errors were minimised during the study by careful sampling, by calibration of the analyzers and methods, and by good planning. Random errors were considered by statistically analysing the results with a risk level of 5%. Nevertheless, during the study a couple of factors occurred that could no longer be eliminated and that could have affected the results:

- Internal recycle pumps in the aeration basins were old, and the actual flow created by them might have been different between the process trains, although the speed was the same. If, e.g. the internal recycle in the primary clarification process train would have been lower than in the equalization/prefermentation process train, it would have improved the phosphorus removal and decreased the denitrification in the primary clarification process train.
- The primary clarification bypass flow was not constant, while the gate valve was easily clogged.
- The reject water from the centrifuges was led to the end of the sand removal basin. The mixing of the reject water in the sand removal was not verified, and thus the reject water might have been unequally distributed between the process trains. The reject water in this case contained high concentrations of soluble organic matter and VFAs.
- The equalization pump dropped the wastewater above the water level, causing some turbulence. A part of the VFA might have evaporated due to this. Samples were taken after the pump, and thus the results might give too low values compared to the VFA concentrations in the equalization/prefermentation basin, but they give correct values of the feed water to the biological process.

4.5 Recommendations for future research

1) Hydrolysis occurring in equalization basins

It was demonstrated that hydrolysis was occurring in the equalization basin and in the primary clarifier. The effects of adding equalization volume to the process on the quality of the organic matter have not been widely studied. Some research can be found on the hydrolysis occurring in the primary clarifiers (Gemaye et al., 2001). It would be necessary to determine in which conditions hydrolysis may occur and whether the hydrolysis could be enhanced. Drewnowski and Makinia (2011) proposed a method for estimating the effect of hydrolysis by parallel measurements with settled wastewater with and without coagulation/flocculation. The pretreatment with coagulation/flocculation removes the colloidal and particulate fraction which corresponds to the SBCOD.

2) Role of the quality of the organic matter in nitrification

As discussed in Section 3.2.5 and in Appendix II, the quality of the organic matter seemed to have an indirect effect on the performance of nitrification. Prefermentation would have changed the quality of the organic matter towards being more readily biodegradable, and this would have benefited nitrification by decreasing the competition for oxygen and space in the aeration basin. It would be necessary to repeat the test in more controlled conditions and thoroughly analyze the composition of the organic matter. Nitrite measurements would be beneficial for determining the nitrifying bacteria responsible for nitrification in the condition with or without prefermentation in order to determine whether the differences are caused by a shift in the bacterial population, as proposed by Dytczak et al. (2007).

3) Distinction between the effects of equalization and prefermentation

As presented in Section 3.1.6, hydrolysis and prefermentation already took place in the equalization basin during the first phase of the study, when only equalization was anticipated. Therefore, it is difficult with our results to differentiate between effects of equalization and prefermentation. It would be necessary to establish the causality of the two pre-treatment processes on the phosphorus release, denitrification, aeration efficiency, sludge settleability, sludge yield and secondary phosphorus release. In order to study this, the study could be repeated with an addition of RBCOD in the unequalized process train in order to create similar characteristics for organic matter in both process trains.

4) Effects of equalization and prefermentation on greenhouse gas emissions

Municipal wastewater treatment causes direct and indirect GHG emissions. Direct emissions occur both from the biological processes of wastewater treatment and biosolids handling in the form of CO_2, CH_4 and N_2O. Indirect carbonaceous emissions are caused by power and heat generation, the manufacturing of chemicals, and the transportation of chemicals and biosolids. Direct GHG emissions caused by equalization are from power generation used for pumping, which in this study was only 0.4% of the total energy consumption at the plant. Based on the results, it seems that a significant reduction in energy consumption and thus in CO_2 emission during sludge handling and transportation could be anticipated. Further research would be needed in order to quantify the energy reductions achieved by equalization.

Emissions of N_2O account for approximately one-third of the total direct emissions within the wastewater treatment sector (Giraldo, 2009). Temperature, DO, nitrites, nitrates, organic carbon limitation, sludge age, pH, ammonium and H_2S are expected to have an impact on the N_2O emissions (Hiatt et al., 2009, Giraldo, 2009, Hanaki et al., 1992). A few studies (Gejlsberg et al., 1998; Ahn et al., 2009) have suggested that mixed liquor ammonium peaks correlate with high N_2O emissions in the aerated zone of the BNR process. An equalization tank can level out the diurnal influent pollutant variation and would limit the potential for N_2O emissions in the aerated zone. Also, Ahn et al. (2009) suggested that N_2O emissions would increase with incomplete nitrification and excess ammonium accumulation. In our study, equalization was shown to be an effective method for increasing nitrification capacity (Appendix II). Thus, it seems that equalization has a potential for minimizing the N_2O emissions, but the role would need to be clarified in further research. Furthermore, prefermentation enhances the denitrification, thus decreasing the risk of N_2O emissions (Johnson and Hiatt, 2009). A future research project would be needed to study the GHG balance of prefermentation.

5 Conclusions

The flow variations in the wastewater plant influent have been shown to have a detrimental effect on the biological treatment process. Flow and load variations have diurnal and weekly patterns caused by the daily rhythm of the connected households and industries. Seasonal flow and load variations are caused by snow melting periods and relatively random rainfall. Also, the plant influent is often lacking readily biodegradable organic matter necessary for BNR. Primary clarifiers in their traditional role of organic load removal are not fully responding to the requirements of the BNR. Therefore, the primary clarifier basin volume could be used in a way that would better benefit the BNR process. In this research project, the existing primary clarifiers at the Pihlajaniemi WWTP in Savonlinna were modified for diurnal flow equalization and prefermentation with raw sludge removal still operational. The performance of the pre-treatment and the BNR process was investigated during an 18-month period. With the experiences of the full-scale test, the four research questions were answered as follows:

How did the pre-treatment perform? The objective in the pre-treatment was to produce readily biodegradable organic matter, to level out the flow variations, and to remove excess organic load before the biological process. This study demonstrated that prefermentation and hydrolysis of the organic matter took place in the pre-treatment basin already in the first phase of the study when no internal sludge recycle was installed. During the first phase, the basin was intended only for flow equalization. The RBCOD and soluble BOD and COD concentrations were significantly higher after the modified pre-treatment than after the primary clarification, but VFA concentrations were in the same range. This would suggest that prefermentation would have not been more efficient in the equalization/prefermentation process train, but most probably part of the VFA produced were lost due to the turbulence caused by the equalization pumping. In the literature, VFA and RBCOD production rates have not been reported in detail for equalization basins, but in this case the rate was in the order of APT tanks. This means that basins used for equalization can significantly modify the quality of the organic matter in the wastewater. On the other hand, the attempt of enhancing the VFA production by adding an internal sludge recycle (Second phase) was not efficient. The production of soluble organic matter was not increased in comparison with values observed during the first phase.

The diurnal flow variations were efficiently levelled out in the existing primary clarifier basin volume. Nevertheless, the available volume was not sufficient for equalization of wet weather flows. On the other hand, it seems advisable to seek other treatment for extreme wet weather flows, because previous studies have suggested that the introduction of the dilute storm water to the BNR process would have a negative effect on the overall treatment performance.

What was the effect on the biological treatment process? The results of the performance of the BNR process showed that the main improvement was observed with nitrification. The removal of ammonium in the equalization/prefermentation process train was 87% on average during the whole 18-month study, whereas in the primary clarification process train only 75% was reached. The improvement could be mainly attributed to the diurnal flow equalization. A part of the observed improvements in the ammonium removal efficiency was also due to the increased heterotrophic assimilation, but the two above could not explain all the observed improvements. In light of our results, it was thus suggested that prefermentation would improve the nitrification activity by ammonification and by decreasing the competition between autotrophic and heterotrophic bacteria. Moreover, the conditions in the equalization/prefermentation process train could have altered the nitrifying population, which would have resulted in a long-term improvement in nitrification capacity. The results on nitrification are of high importance because the effects of dynamic influent and feed water characteristics on nitrification have not been widely studied, especially not in full scale. This study gives some indication of how the improved nitrification achieved by equalization could be taken into account in plant design.

Furthermore, this study confirmed the earlier findings that stable flow conditions improve the removal of organic matter and SS mainly through a better hydraulic control of secondary clarifiers. Denitrification was shown to be improved by the increased amount of soluble organic matter. The phosphate release was also enhanced during a warm period, but the effects of the prefermentation on biological phosphorus removal could not be concluded infallibly due to the potential secondary release of phosphorus in the secondary clarifiers, possible inhibition of enhanced biological phosphorus removal, and the unclear role of chemical phosphorus precipitation. An important finding during the study was that parts of the sludge blanket in the secondary clarifier might be anaerobic, although the return sludge remained anoxic.

What was the effect on sludge production and energy consumption? The equalization and prefermentation improved the sludge settling characteristics. It is suggested that this was due to a more complete biological degradation process in the equalization/prefermentation process train. Moreover, sludge production and yield were decreased. This was attributed to the production of forms of COD compounds with lower yield characteristics during the prefermentation and to the stable process conditions offered by equalization enhancing the biological degradation. The aeration efficiency was improved by equalization, although the air requirement remained the same due to the increased load. The raw sludge removal in the equalization/prefermentation basin was not compromised due to the modified operation. Furthermore, based on a theoretical assessment, equalization seems to have potential for reducing energy consumption of the wastewater treatment.

How could the need for equalization and prefermentation be assessed in advance? It has been difficult to assess the effects of influent variations, because until now no commonly accepted method for measuring the magnitude of flow variations has existed. This thesis introduces a coefficient of flow variation that can be used for this purpose. Moreover, it was demonstrated that by analyzing the influent wastewater quality, valuable information for assessing the performance of the prefermentation could be gathered and an approximate picture of reactions taking place in the sewer network could be drawn. It was observed that during dry weather flows, a significant amount of soluble organic matter was produced in the sewer system. During summertime, significant removal of organic matter and of nitrogen was also occurring in the sewer system, and thus the VFA forming potential was higher during the cold period. The seasonal variation in the influent quality should be taken into account in the operation of prefermentation.

It can be concluded that the use of the existing primary clarifier volume for diurnal flow equalization and prefermentation is a feasible solution for the improvement of the BNR process performance. This process modification could be widely implemented in Finland because a majority of middle-sized and big WWTPs have primary clarifiers. Nevertheless, it is not possible to rank equalization and prefermentation, because many of the improvements could not be attributed clearly to one or another. On the other hand, it seems that hydrolysis is anyway taking place in equalization basins without an internal sludge recycle.

The economical savings achieved by these modifications are important. The diurnal flow equalization can significantly improve nitrification, which is usually the limiting process in the biological wastewater treatment, and thus enables a reduction in the aerated process volume. The external carbon source can be replaced by using the wastewater's own carbon more efficiently through prefermentation. Moreover, the results of this study are not only helpful in treatment plants where flow equalization or prefermentation is used, but the findings may also help the designers and operators to better understand the problems occurring at the plants, which in many cases are related to the quality of the influent water. In any case, it can be emphasized that other possibilities of levelling out the flow and load variations at the plant (for example, reject and wash water recycles, equalization capacity in the sewer system, and storage for septic tank sludges) can offer significant improvements and should be considered carefully. Furthermore, the variability of the conditions occurring at the plant should be reflected in the process design and operation strategies, and easily modifiable pre-treatment units should be further developed and implemented.

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The flow and load variations in the wastewater plant influent complicate the operation of the biological treatment process and harm the process performance. The daily discharge rhythm of the connected households and industries create diurnal and weekly variations. Seasonal variations are caused by the snow melting period and relatively random rainfall. Moreover, when the biological nutrient removal (BNR) process is implemented in the plant, the plant influent is often lacking readily biodegradable organic matter. The organic matter in the influent is often in a slowly biodegradable form and not accessible to the bacteria. Readily biodegradable organic matter can be produced by prefermentation.

This thesis focuses on these two challenges of the biological wastewater treatment – flow variations and the quality of the organic matter. The performance of the pre-treatment and the BNR process was investigated during an 18-month period at the Pihlajaniemi WWTP in Savonlinna.

Aalto-DD 76/2013

ISBN 978-952-60-5152-9
ISBN 978-952-60-5153-6 (pdf)
ISSN-L 1799-4934
ISSN 1799-4934
ISSN 1799-4942 (pdf)

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