DOKUZ EYLÜL UNIVERSITY
GRADUATE SCHOOL OF NATURAL AND APPLIED SCIENCES

MANAGEMENT OF PHOSPHORUS REMOVAL IN MUNICIPAL WASTEWATER TREATMENT PLANTS

by
Tolga TUNÇAL

March, 2008
İZMİR

MANAGEMENT OF PHOSPHORUS REMOVAL IN MUNICIPAL WASTEWATER TREATMENT PLANTS

A Thesis Submitted to the
Graduate School of Natural and Applied Sciences of Dokuz Eylül University
In Partial Fulfillment of the Requirements for the Degree of Doctor of Philosophy in Environmental Engineering, Environmental Technology Program

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Ph.D. THESIS EXAMINATION RESULT FORM

We have read the thesis entitled **"MANAGEMENT OF PHOSPHORUS REMOVAL IN MUNICIPAL WASTEWATER TREATMENT PLANTS"** completed by **TOLGA TUNÇAL** under supervision of **Prof. Dr. AYŞEGÜL PALA** and we certify that in our opinion it is fully adequate, in scope and in quality, as a thesis for the degree of Doctor of Philosophy.

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ACKNOWLEDGMENTS

I would like to express my gratitude to my advisor, Prof. Dr. Ayşegül Pala for her guidance, encouragement, support and valuable advices during this study.

My deepest thanks go to Prof. Dr. Orhan Uslu for his guidance, advices and expertise throughout the study. Without his knowledge, this study would not have been successful. I am sincerely grateful to my thesis committee member, Prof. Dr. Filiz Küçüksezgin, for her technical supports during my thesis development.

I am also grateful to Environmental Engineer Faruk İşgenç, Manager of the Çiğli WWTP and laboratory personnel İlhan Çiçek, Aytekin Parga and Aysun Yıldırım for their technical supports.

Finally, I would like to express my deepest appreciation to my wife, Perihan Tunçal and my daughter, Deren Tunçal for their endless patience, support and moral motivation during my thesis studies.

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ABSTRACT

As a result of uncontrolled discharge of wastewaters containing high levels of phosphorus to the receiving bodies could create adverse effects on water quality. Therefore phosphorus should be eliminated prior to discharging into aquatic environment together with carbon and nitrogen to reduce eutrophication risk. Control of effluent phosphorus level by enhanced biological methods became a standard wastewater treatment application due to its high wastewater purification efficiency and its enhancing effect on overall biological treatment stages. In enhanced biological phosphorus removal (EBPR) processes, carbon, nitrogen and phosphorus could be removed simultaneously with higher efficiencies as compared to the conventional biological treatment methods.

Another common method applied in wastewater treatment plants (WWTPs) for removal of phosphorus is chemical precipitation method in which aluminum and ferrous salts, lime were used as coagulants. Although these methods are not able to remove phosphorus completely from the theoretical aspect, significant phosphorus removal efficiencies could also be achieved. However chemical phosphorus removal methods could increase sludge production rate significantly in WWTPs. Investigations indicated that characteristics of this chemical sludge are very different from typical activated sludge. It was also reported that sludge treatment applications including thickening – dewatering, anaerobic digestion, thermal drying and incineration could be more complex and costly as compared to the non-chemical sludge.

In this study, fundamental characteristics of EBPR were investigated in a large wastewater treatment plant by full-scale and laboratory scale methods. The process configuration in which investigations were carried out was very similar to 5-Stage Modified Bardenpho process. To obtain accurate results mass balances were

established around biological treatment units performing detailed influent and effluent characterization. In addition to the wastewater characterization, phosphorus content of both activated sludge and phosphorus accumulating organisms (PAOs) were examined under variable nutrient loading ratios and variable operational conditions. Anaerobic phosphorus release rate, soluble substrate utilization rate, anoxic phosphorus uptake rate, simultaneous denitrification rate and aerobic phosphorus uptake rate were also determined by laboratory-scaled batch tests in addition to the full-scale investigations. Importance of particulate substrate forms in EBPR was also studied using both full-scale and batch-scale methods.

Results of the investigations proposed that EBPR method could be the main phosphorus management method in WWTPs. However it was also observed that the stability of the EBPR processes could not be maintained continuously since the process influences by several wastewater and operational parameters. Therefore EBPR processes could be supported by chemical phosphorus removal methods to achieve target effluent phosphorus level.

It was also demonstrated that COD/TP ratio of the influent is very important for the optimization of the EBPR processes. Obtained results indicated that COD/TP ratio of 65 could be required for the domination of phosphorus accumulating microorganisms (PAOs) that are perquisites EBPR processes in the scale of İzmir WWTP. Serious adverse effects of electron acceptors, present in both influent and return sludge line, on EBPR performance were also determined. In the scope of this thesis, several operational strategies for the optimization of the EBPR plant were also developed.

Keywords: Biological phosphorus removal (EBPR), phosphorus accumulating bacteria (PAOs), mass balance, anaerobic hydraulic retention time (HRT_a), electron acceptor, activated sludge, full-scale, large EBPR wastewater treatment plant (LWWTP)

EVSEL ATIKSU ARITMA TESİSLERİNDE FOSFOR ARITIMININ YÖNETİMİ

ÖZ

Yüksek konsantrasyonlarda fosfor içeren atıksuların kontrolsüz şekilde alıcı ortama deşarjı edilmesi sonucunda su kalitesinde önemli ölçüde bozulmalar oluşabilmektedir. Su kalitesinin korunması ve alıcı ortamlarda ötröfikasyon riskinin azaltılması için karbon ve azot gibi nütrientlerin yanı sıra fosforun da arıtılması gerekli olabilmektedir. Arıtılmış su fosfor seviyesinin kontrolünde geliştirilmiş biyolojik yöntemlerin kullanımı yüksek nütrient giderim veriminin yanı sıra, olumlu yönde tüm biyolojik arıtma süreçlerini de etkilemesi nedeniyle günümüzde rutin bir atıksu arıtım uygulaması haline gelmiştir. Geliştirilmiş biyolojik fosfor giderimi (GBFG) süreçlerinde, konvansiyonel arıtma yöntemlerinden farklı olarak, karbon, azot ve fosforun eşzamanlı olarak, yüksek verimde giderilmesi de mümkün olmaktadır.

Atıksu arıtma tesislerinde fosfor giderimi amacıyla yaygın olarak kullanılmakta olan bir diğer yöntem ise demir ve alüminyum tuzları, kireç gibi kimyasalların kullanıldığı çökeltme işlemleridir. Bu yöntemler fosforu, teorik olarak tamamen gideremmemelerine karşın oldukça önemli verimler de sağlayabilmektedir. Ancak bu kimyasal çökeltme işlemleri, arıtma tesislerinde oluşan çamur miktarını oldukça arttırabilmektedir. Yapılan araştırmalar bu çamurların niteliğinin, klasik aktif çamurundan oldukça farklı olduğunu göstermiştir. Bunun ötesinde, söz konusu kimyasal arıtma çamurlarının; yoğunlaştırma – susuzlaştırma, anaerobik çürütme, termal kurutma ve yakma gibi temel çamur arıtım işlemlerinin, aktif çamura göre daha karmaşık ve daha maliyetli olduğunu göstermiştir.

Bu tez çalışmasında büyük ölçekli bir atıksu arıtma tesisinde GBFG mekanizması hem saha ölçekli hem de laboratuar ölçekli çalışmalarla incelenmiştir. İncelemeler 5-kademeli Modifiye Bardenpho prosesine oldukça benzerlik gösteren bir proses yapısında gerçekleştirilmiştir. Giriş ve çıkış atıksuyu, nütrientler ve bunların çeşitli formları için karakterize edilmiş ve biyolojik arıtma ünitelerini kapsayan kütle

dengeleri oluşturulmuştur. Atıksu karakterizasyonuna ilave olarak aktif çamur fosfor içeriği, fosfor depolayan bakterilerdeki (FDB) hücre içi fosfor içeriği ve bu bakterilerin aktif çamurdaki oranı, değişken nütrient yükleri ve işletme parametreleri altında irdelenmiştir. Saha ölçekli araştırmaların yanı sıra, laboratuar ölçekli yöntemlerden de faydalanılarak, aktif çamurun oksijensiz ortamda fosforu salım hızı ve çözünmüş formdaki besin maddelerini tüketim hızı, anoksik ortamda fosforun bakteri bünyesine geri alım hızı ve eş zamanlı olarak meydana gelen nitrat asimilasyonu, oksijenli ortamda fosforun bakteri bünyesine alım hızı tespit edilmiştir. Partikül formda bulunan besin formlarının BFG süreci açısından önemi hem saha hem de laboratuar ölçekli kesikli deneyler ile irdelenmiştir.

İncelemeler neticesinde, evsel atıksu arıtma tesislerinde ana fosfor giderim yönteminin GBFG olabileceği tespit edilmekle beraber bu sistemlerin ham atıksu karakteristiğine ve işletme değişkenlerine son derece bağlı olduğu için stabilitesinin süreklilik arz edemeyebileceği gözlemlenmiştir. Bu neden dolayı, GBFG yöntemlerinin, kimyasal fosfor giderim yöntemleri ile desteklenerek, arıtılmış suda istenilen fosfor konsantrasyonu hedefine ulaşılabileceği sonucuna varılmıştır.

GBFG süreçlerinin performansının optimizasyonu için ham atıksu KOİ/TP oranının son derece önemli olduğu, bu süreçlerin temel özelliği olan, bünyelerinde yüksek oranda fosfor depolayabilen bakterilerin (FDB) aktif çamur sisteminde baskın hale gelebilmeleri için, İzmir Atıksu Arıtma Tesisi ölçeğinde, KOİ/TP oranının 65'in üzerinde olması gerektiği saptanmıştır. Geri devir çamur hattında ve giriş suyunda bulunan elektron alıcılarının (EA) sistem performansını önemli ölçüde bozabileceği belirlenmiştir. GBFG süreçlerinin optimizasyonu için çeşitli işletme senaryoları geliştirilmiştir.

Anahtar Kelimeler: Geliştirilmiş Biyolojik Fosfor Giderimi (GBFG), fosfor depolayan bakteriler (FDB), kütle dengesi, oksijensiz ortam hidrolik alıkonma süresi (HAS), elektron alıcılar (EA), aktif çamur, tam ölçekli çalışma

CONTENTS

Page

CHAPTER ONE
INTRODUCTION

1.1 Introduction

İzmir bay is one of the great natural bays of the Aegean sea. Total surface area of the bay is 500 km^2 and total water volume is 11.5 billion m^3. The bay could be examined in 3 main sections according to the physical characteristics of the different water masses. These sections are named as outer, middle and inner bay. The depth of the water decreases from the outer bay to the inner bay and the average water depth in outer bay is 70 m (Kucuksezgin et. al., 2005). Scientific investigations indicated that eutrophication of the inner bay was a serious problem lasting whole year and red tide events are becoming more frequent (UNEP, 1993; Kontas et. al., 2004).

It was also found that inner bay phosphate concentration was higher than the values measured in clean waters. The origin of this higher level of phosphate concentration was domestic wastewaters. The atomic ratio of TNO_x to phosphate in outer bay was reported between 1.8 to 27 and 0.02 to 54 in the middle and inner bay. It was also reported that the observed average N/P ratios were lower than optimal growth requirement (N/P=15/1) in conformity with Redfield's ratio (N/P=16) in the bay. According to the measured N/P ratios, nitrogen is limiting nutrient in the İzmir bay. However phosphorus could also became a limiting nutrient in the summer period due to *cyanobacteria* activity. Scientific investigations indicated that pollution level in the outer bay was not significant but eutrophication of the inner bay has already begun and could be spreading to the outer part of the bay (Kucuksezgin et. al., 2005).

To prevent discharge of untreated wastewaters into the bay, İzmir WWTP was taken into operation in early 2000. The plant was designed to treat both domestic and pre treated industrial wastewaters collecting from the İzmir metropolitan area. Since previous scientific investigations indicated that both Nitrogen (N) and Phosphorus (P) concentrations of the sea were critical level with respect to eutrophication

problem, the plant design was performed for combined removal of N, P in an activated sludge process following adequate physical treatment including fine screens, aerated grit removal chambers and circular primary sedimentation tanks. The initial average design capacity of the plant is 604,800 m^3/d.

Phosphorus (P) is an essential nutrient for all life forms. It is also one of the limited and non-renewable natural resources therefore it should be recovered from wastewater. Furthermore treated wastewater containing high level of P could cause serious problems associated with eutrophication in receiving water bodies (Janssen et al., 2002; WPCF, 1991).

Removal of nutrients by biological methods is cost effective and environmentally sound alternative to the chemical treatment of wastewater (Osee et al., 1997). Although controlling nutrient discharges to receiving bodies by biological methods have many advantages, several disadvantages such as dependence on wastewater composition, lower stability and flexibility, influence on the sludge volume index (SVI) and P release in the sludge treatment, should also be considered. One of the most important advantages of EBPR over CPR is occurrence of no chemical sludge. CPR leads an increase in total sludge production because of improved suspended and dissolved solids removal efficiency, formation of metal – hydroxide precipitants etc. Increased sludge production rate also leads to higher costs of sludge treatment, disposal and final management (Janssen, Meinema, van der Roest, 2002).

It was also reported that chemical phosphorus precipitation is a very critical from biochemical point of view. When the dosage is too high the orthophosphate will be bound chemically and will not be available for PAOs for energy conversation. As result of that PAOs could not utilize simple substrate forms in the anaerobic period and finally, they could be washed out of the activated sludge system. Insufficient selection of PAOs will definitely results an increase of effluent phosphate concentration that will then require higher amount of metal salt dosage. In addition to increased sludge volume, dewatering characteristics of the sludge could be deteriorated as a result of chemical phosphorus removal methods. In contrary to

chemical methods, phosphate is preserved as potassium or magnesium phosphate (polyphosphate) without adhered water, resulting in a good dewaterable product (Janssen, Meinema, van der Roest, 2002).

Use of metal salts and lime for phosphorus removal not only negatively affects sludge dewatering characteristics but also influence other sludge disposal unit operations including anaerobic sludge digestibility, thermal drying and incineration of the sludge (USEPA, 1987). Since chemically removed sludge contains more adhered water, moisture content is higher than the non-chemical sludge. Therefore in combination of increased sludge volume and higher sludge moisture content, energy requirement that is one of the most determinative factors in operational costs will be significantly higher as compared to non-chemical sludges from the point of sludge mixing, pumping, and heating (anaerobic digestion, thermal drying and incineration).

Another disadvantage of CPR is increased salinity and conductivity level in effluent originating from remaining negative ions (chloride and sulfide). In addition to increased salinity, effluent total dissolved solids (TDS) concentration could increase due to impurities present in the metal salts (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

Addition of metal salts could destroy the wastewater buffer capacity that could result with inhibited or poorer nitrification efficiency especially for wastewater with week buffer capacity. Dosing of chemicals will remove available COD not only for PAOs but also for the denitrifiers that means lowered denitrification efficiency. It could be concluded that CPR processes could deteriorate overall nitrogen removal efficiency (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

Phosphorus is one of the non-renewable and limited natural sources. Moreover, increased industrial and agricultural phosphorus demand increased the market price of the phosphorus (Isherwood, 2000). Investigations showed that recovery of chemically precipitated phosphorus is not possible or at least not feasible (Donnert and Salecker, 1999a; Donnert and Salecker, 1999b). In contrary, recovery of

phosphorus from wastewater with an efficiency range of (10% -80%) is only possible with EBPR processes in which sludge became rich in phosphate (Strickland, 1999; Gaterell et al., 2000; Jeanmaire & Evans, 2001). It was also reported that fertilizer effect of EBPR sludge is attractive resource for agricultural purpose (Meinema, van der Roest, 2002).

As it could be seen above given literature survey use of biological methods over chemical methods for removal of phosphorus from wastewaters have many advantages. However many scientific investigations were also demonstrated that EBPR removal processes are highly dependent upon influent wastewater characteristics such as; COD/TP ratio, pH and temperature etc. In addition, sludge handling systems should be carefully designed and well operated to prevent phosphorus releases back to the plant (Meinema, van der Roest, 2002; Metcalf & Eddy, 2003). Therefore the stability of the EBPR process should be supported by CPR process to provide effluent discharge standards. However these systems should include fully automatic control systems

Mechanism of EBPR removal is based on exposing microorganisms to an anaerobic/aerobic sequence that results with selection of P accumulating microorganisms (PAOs) in the activated sludge culture. Preferential selection of PAO in system is attributed to energy conversion ability of these organisms from storage of simple carbohydrates and release of P in the anaerobic zone of the EBPR processes. In these assimilative reactions energy is derived from hydrolysis of intracellular poly-P reserves (Comeau et al., 1986; Mino et. al. 1987; Wentzel et. al, 1991). Stored carbohydrates are utilized to generate required energy for reproduction of new cells and restoring depleted poly-P reserves using electron acceptors either in the form of DO or nitrate in the aerobic zone (Kuba et. al., 1996; Lee et. al., 2003).

EBPR process could be characterized with consumption of readily biodegradable soluble substrates (rbsCOD) and release of P into liquid interface in the anaerobic zone and the uptake of excess amount of P into microbial cell in the following aerobic zone.

Establishing a successful EBPR process mainly depends on accurate characterization of wastewater with respect to both concentrations and loading rates. Fluctuations in hydraulic and organic loading rates should be equalized and overloading of the biological stages should be prevented (Shehab et al., 1996). In addition to physical characteristics, chemical composition of wastewater should also be well defined with respect to nutrients as well as their various forms. The most important and deeply investigated chemical parameter in EBPR is COD and its fractions. Especially, availability of rbsCOD in the anaerobic zone is one of the essential considerations.

It was reported that at least 20 mg of rbsCOD as acetic acid (Abu-ghararah and Randall, 1991), 50 mg as COD (Ekama and Marais, 1984) and 7–9 mg as VFAs (Barnard, 1993) are required to remove 1 mg of P. Ratio of COD/TP is an useful definition to predict effluent quality of EBPR process. It was reported that COD/TP ratio 40:1 are required to achieve effluent total P concentration of 1 mg/l or less (Randall et al., 1992). Since rbsCOD/COD ratio shows drastic changes regionally, use of rbsCOD/TP over COD/TP ratio as an indication of stabile EBPR seems to be more reliable due to insufficient selection of PAOs in absence of sufficient soluble substrate forms in anaerobic the zone.

Composition of influent that could be characterized by the main parameters pH, temperature, conductivity, DO, etc., affect EBPR. Investigation results indicated that substrate consumption and P release rates were adversely affected by acidic pH in the anaerobic zone; in addition, alkaline pH levels resulted with inhibition of acetate uptake but stimulated P release (Liu et al., 1996). Temperature is another critical parameter in EBPR that should be evaluated together with the sludge age (SRT or θ_c). An experimental study indicated that EBPR performance was optimal at SRT ranges of 16-24 days for 5 °C and 12-17 days for 10 °C. It was concluded that EBPR microorganisms are capable to alter their biochemical metabolism based on environmental conditions (Erdal et al., 2003).

Another important concern in maintaining successful EBPR is the presence of electron acceptors in the anaerobic zone. Concentrations of DO or nitrate in the anaerobic zone should be minimized in order to obtain high EBPR removal efficiency. These electron acceptors are mainly carried to the anaerobic zone by influent and recycle streams in the form of both nitrate and DO. Organic matter that is required for PAOs will be oxidized by ordinary heterotrophic bacteria in the presence of electron acceptors. P release in anaerobic phase was inversely proportional to the amount of nitrate present when excess substrate was available. Denitrification of nitrate in the anaerobic zone had the effect of reducing the availability of substrate for P release (USEPA, 1987).

Another serious adverse effect of electron acceptors on EBPR efficiency was explained using recent molecular identification techniques called fluorescence in situ hybridization (FISH) analysis. This analysis combined with laser scanning microscopy of EBPR system showed that change of the electron acceptor from oxygen to nitrate resulted in a shift in bacterial population from alpha subclass to filamentous, beta subclass bacteria within two weeks (Falkentoft, 2002). Occurrence of filamentous bacteria was observed as a serious problem during the investigation period; this was supporting the above given results due to electron acceptor input to the anaerobic zone.

At the first part of the thesis, simultaneous effect of variable HRT_a and presence of electron acceptors in anaerobic zone was investigated. To obtain comparable and accurate results, mass balances for P, sCOD, Total Nitrogen (TN), Nitrate and DO were set up around the biological treatment units including the anaerobic, anoxic, aerobic zones and the recycle stream. Flow data including, inflow, external -internal sludge recirculation and sludge withdrawal rates were monitored and recorded continuously by computer aided, online measurement system during the monitoring period. In addition to the EBPR investigation results, wastewater characterization, general biological nutrient removal (BNR) performance including COD and TN removal of the WWTP were reported as well.

At the second part of the thesis, EBPR process, loading under different rbsCOD/TP ratios were investigated. In order to evaluate EBPR process accurately, influent and effluent wastewater were characterized and mass balances for PO_4-P, sCOD, Total Nitrogen (TN) and Nitrate were established around biological treatment units. Influent and effluent of anaerobic, anoxic and aerobic zone recycle stream and effluent of final sedimentation tanks were selected as monitoring points. All required data for accurate evaluation of the EBPR process including environmental and operational factors such as pH, temperature, mixed liquor volatile suspended solids (MLVSS) concentrations, hydraulic retention times in biological treatment units and SRT were recorded during the monitoring period. Flow data including inflow, external -internal sludge recirculation and sludge withdrawal rates were monitored and recorded continuously by computer aided, online measurement system during the monitoring period. All experimental results were evaluated considering theoretical background of EBPR.

At the third stage of the thesis previously obtained full scale measurement results and the established mass balances were used for characterization of the activated sludge community in terms of PAOs and non-PAOs cell synthesis and phosphorus enrichment of the microorganisms. In addition to previously obtained full scale measurement results and the established mass balances, biodegradable COD (bCOD) and particulate biodegradable COD (pbCOD) that were determined using the BOD data were combined to estimate PAOs and non-PAOs cell synthesis and phosphorus enrichment of the microorganisms. Additionally behavior and microbial composition of the activated sludge under variable organic and hydraulic loading rates were investigated.

In the fourth phase of the thesis importance of biodegradable materials in EBPR was investigated. Experimental studies were carried out in a two separate full-scale treatment lines working as parallel. In one line, inflow was fed to PS prior to EBPR process and in the second line; inflow was fed directly to the anaerobic tank of the EBPR process. The EBPR characteristics and SVI values of these treatment lines were compared with each other. Influent and effluent of both two treatment lines

were characterized by full scale measurements. Important process variables including sludge age, MLVSS, internal and external flow rates, SVI were monitored continuously. Obtained results were evaluated by both statistically and graphically using the reviewed theoretical background. In addition to the full-scale studies, activated sludge samples that were obtained from these two treatment lines were investigated deeply using laboratory scaled batch tests to compare their EBPR characteristics.

In the last phase of the thesis, all of the investigation results were combined together with reviewed literature background to develop management strategies for establishment of a reliable phosphorus removal method in WWTPs. In this part of the study, several alternatives to improve EBPR performance were also discussed deeply and reliability of the proposed methods were demonstrated clearly using the results obtained from the investigations that have been conducted for nearly two years in a full scale WWTP. In this chapter mass balance results were demonstrated as loading rate and internal recycle of the phosphorus in the WWTP were defined graphically. Using these mass balance results, operational strategies and several protective actions were implemented to improve EBPR process. Since performance of the EBPR processes could be variable under different feeding and operational conditions, application of CPR in combined with EBPR was also discussed under this chapter.

CHAPTER TWO
THEORETICAL BACKGROUND

2.1 Impact of nutrient control in aquatic environments

Eutrophication is the term used to define high biological productivity in a body of water. Eutrophication generally originated from nutrient uptake by phytoplankton and other aquatic growths. These resultant organisms and plants eventually die and settle to the bottom of the receiving water body. These settled death organisms are decomposed by biological activity and nutrients are released back to the liquid phase. This nutrient chain continues during the lifetimes of tens or hundreds of thousands of years. These lifetimes could be evaluated in three main distinct phases. The first is the "*oligotrophic phase"* where biological productivity is low because of low nutrient loadings. As nutrient loadings increase the "*mesotrophic phase"* in which a greater biological productivity develops. With excess amount nutrient loading form external sources and internal nutrient recirculation the "*eutrophic phase"* starts. In this phase the biological productivity is much greater than the other first two phases. Therefore the main purpose of the nutrient control is to reduce the external loading rate and minimize eutrophication process (WPCF, 1983).

Several parameters could determine the eutrophic level of a water body including; standing crop of phytoplankton, level of chlorophyll a, volume of algae, level of oxygen production, level of oxygen depletion, Secchi disk readings. Investigations carried out in worldwide have indicated that untreated domestic and industrial wastewaters, drainage from agricultural and urban areas could distribute certain quantity of nutrients that may increase the biological productivity in receiving aquatic environments (WPCF, 1983).

The primary nutrients required for the algae growth was listed in Table 2.1. The primary source of the carbon is carbon dioxide. As for all green plants, nitrogen is derived from ammonia and nitrate. Phosphate ion is the soluble source of phosphorus and dissolved silicates are the source of silicon. In addition, nitrogen-fixing algae are

able to use dissolved nitrogen gas in water when ammonia and nitrate is limited. Further studies revealed that control of carbon was not reasonable due its large stores of half-bound carbon dioxide that exist in most natural waters in the form of bicarbonates (alkalinity) that are available for algae reproduction. Another main aspect recognizing the control of carbon is originated from carbon dioxide which is derived from bicarbonates. Actually rise of pH due to the shift to carbonate and hydroxyl ions thus making carbon dioxide more readily absorbable from the atmosphere.

Table 2.1 Primary nutrients for the production of algae (based on dry mass)

Nutrient	**Primary Nutrients %**
Carbon	35 - 50
Nitrogen	3 - 10
Phosphorus	0.5 - 1.0
Silicon	0.1 - 14

Nitrogen in the algae biomass varies from 3% to 10% mainly in the form of proteins. While green algae needs smaller amounts of nitrogen for reproduction blue-green algae needs larger amounts of Nitrogen. In addition to external nitrogen sources, some blue-green algae could fix elemental nitrogen dissolved in water. Several studies showed that nitrogen could became limiting in the summer growing season at a limiting concentration of 0.05 mg/L.

Although phosphorus content of algae cells occurs in small amounts ranging from 0.5 to 1.0%, it has been proved to be a limiting factor in the growth of algae in many cases. A value of less than 0.005 mg/l or 5 µg/L in the orto form is recognized as a lower growth limiting concentration. In many cases control of phosphorus is absolutely essential because, when nitrogen becomes limiting, any excess of phosphorus could support growth of nitrogen-fixing blue-green algae.

One undebatable fact explaining phytoplankton and aquatic plants was described by Birge & Juday in 1922. Using the well-known method of Justus von Liebig, they postulated the nutritional requirements for these aquatic organisms summarized in Table 2.2.

Table 2.2 Nutrient levels of some typical green algae, blue-green algae and rooted aquatics.

	Percent, dry mass			
Plant	**Carbon**	**Nitrogen**	**Phosphorus**	**N/P**
Blue-green algae				
Anabaena	49.7	9.43	0.77	12/1
Aphanizomenon	47.7	8.57	1.17	7/1
Mirocystis	46.5	8.08	0.68	12/1
Green algae				
Cladophora	35.3	2.30	0.56	4/1
Pithophora	35.4	2.57	0.30	8/1
Spirogyra	42.4	3.01	0.20	15/1
Rooted aquatics				
Elodea	-	2.10	0.14	15/1
Lobelia	-	1.89	0.16	12/1
Potomogeton	-	3.19	0.30	11/1

2.1.1 Water Quality in İzmir Bay

İzmir bay is one of the great natural bays of the Aegean sea. Total surface area of the bay is 500 km^2 and total water volume is 11.5 billion m^3. General layout and sewer network of the İzmir city is demonstrated in the Figure 2.1. The bay could be examined in 3 main sections according to the physical characteristics of the different water masses. These sections are named as outer, middle and inner bay. The depth of the water decreases from the outer bay to the inner bay and the average water depth in outer bay is 70 m (Kucuksezgin, Kontas, Altay, Uluturhan, Darılmaz, 2005).

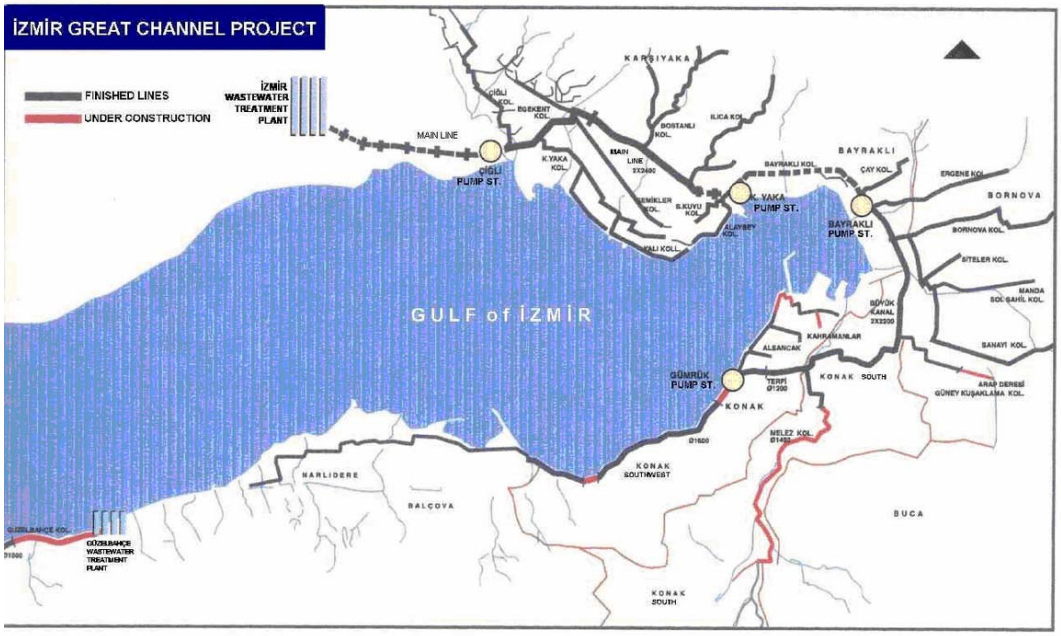


Figure 2.1 Layout of the İzmir Gulf , main sewerage network and central WWTP

Scientific investigations indicated that eutrophication of the inner bay was a serious problem lasting whole year and red tide events are becoming more frequent (UNEP, 1993; Kontas et. al., 2004).

Nutrient levels measured in1996-2003 revealed that winter-autumn TNO_x-N concentrations were higher than spring-summer seasons that could be explained with lowered phytoplankton nutrient uptake. High chlorophyll-a concentrations were also measured in the outer bay during the winter season and this gradually high chlorophyll-a concentration were most probably originated from the heavily polluted Gediz river flowing into outer bay (UNEP, 1993).

In the middle and inner bay nutrient and chlorophyll levels were higher than outer bay. Maximum levels of phosphate and nitrate+nitrite in the inner bay were measured in the summer and autumn period attributed to higher bacterial decomposition activities. It was also found that inner bay phosphate concentration was higher than the values measured in clean waters. The origin of this higher level of phosphate concentration was domestic wastewaters. The atomic ratio of TNO_x to phosphate in outer bay was reported between 1.8 to 27 and 0.02 to 54 in the middle

and inner bay. It was also reported that the observed average N/P ratios were lower than optimal growth requirement (N/P=15/1) in conformity with Redfield's ratio (N/P=16) in the bay. According to the measured N/P ratios, nitrogen is limiting nutrient in the İzmir bay. However phosphorus could also became a limiting nutrient in the summer period due to *cyanobacteria* activity. Scientific investigations indicated that pollution level in the outer bay was not significant but eutrophication of the inner bay has already begun and could be spreading to the outer part of the bay (Kucuksezgin, Kontas, Altay, Uluturhan, Darılmaz, 2005).

2.2 Chemical phosphorus removal (CPR)

Phosphorus could be removed from wastewaters by chemical, biological and physical methods. In chemical methods a variety of metal salts are used for removal of phosphorus and most commonly used chemicals consists of lime, aluminum sulfite (alum) and ferric chloride. EBPR is based on the uptake of phosphorus beyond its normal microbial growth requirements by modified activated sludge processes. Physical removal processes are used to intercept phosphorus from aquatic environment and most commonly applied physical methods are ultrafiltration, reverse osmosis and ion exchange (WPCF, 1983; USEPA, 1987; Metcalf & Eddy, 2003). Selection of these models should be based on capital cost, local chemical costs, reliability of selected method, impacts on the other unit operations including sludge dewatering and final sludge disposal and ultimate disposal of the intercepted phosphorus to prevent accidental reentry of the phosphate back to the system (WPCF, 1983).

Mineral salts are used to precipitate phosphorus and the reactions between phosphorus and metal salts are complex. Aluminum compounds are one of the most preferred mineral salts and especially alum ($Al_2(SO_4)_3 \cdot 14H_2O$) is the most common chemical applied in full-scale municipal WWTPs. A general reaction illustration could be as follows;

$$Al^{+3} + PO_4^{-3} \rightarrow AlPO_4 \downarrow$$

The reaction of alum with phosphorus could be described by:

$$Al_2(SO_4)_3 \bullet 14H_2O + 2PO_4^{-3} \rightarrow 2AlPO_4 \downarrow +3SO_4^{-2} + 14H_2O$$

One mole (594 g) of alum reacts with 2 moles (190 g) of phosphate containing 62 g phosphorus to form 2 moles (244 g) of $AlPO_4$. Thus the weight ratio of alum to phosphorus is 594 g to 62 or 9.6:1. The optimum pH for phosphorus removal aided by alum is in the range of 5.5 – 6.5. In practice the pH is adjusted by adding excess amounts of alum and pH adjustment by acids is not preferred (USEPA, 1987).

Iron salts are most commonly preferred minerals used as phosphorus precipitant in municipal wastewater treatment. Both ferrous (Fe^{+2}) and ferric (Fe^{+3}) ions could be used in the form of ferric chloride, ferrous chloride, ferric sulfite and ferrous sulfite. A typical reaction between ferric chloride and phosphate could be approximated as follows;

$$FeCl_3 + PO_4^{-3} \rightarrow FePO_4 \downarrow +3Cl^-$$

The molar ratio of Fe to P is 1:1 as could be seen the above given reaction. 162.3 g of $FeCl_3$ will react with 95 g of PO_4 to form 150.8 g of $FePO_4$. Stoichiometric weight ratio of Fe:P is 1.8:1. As with alum, the reaction mechanism is more complex than the above given equation (USEPA, 1987).

The reaction between ferrous salts and phosphate could be expressed as follows;

$$3FeCl_2 + 2PO_4^{-3} \rightarrow Fe_3(PO_4)_2 \downarrow +6Cl^-$$

$$3FeSO_4 + 2PO_4^{-3} \rightarrow Fe_3(PO_4)_2 \downarrow +3SO_4^{-2}$$

Ferrous chloride and ferrous sulfate are available as byproducts of steelmaking and these solutions may contain large quantities of free hydrochloric or sulfuric acid

which could cause serious destruction of alkalinity and depression of pH as demonstrated in the following reaction order;

$$3FeCl_3 + 3HCO_3^- \rightarrow FeOH_3 + 3CO_2 + 3Cl^-$$

The optimum pH for ferric ion ranges 4.5-5.0 and for ferrous ion, optimum pH is 8.0.

Table 2.3 Chemical features of the several precipitants (Metcalf & Eddy, 2003

				Availability	
Chemical	**Formula**	**Molecular weight**	**Equivalent weight**	**Form**	**Percent**
Alum	$Al_2(SO_4)_3.18H_2O$	666.5	114	Liquid	8.5 (Al_2O_3)
				Lump	17 (Al_2O_3)
	$Al_2(SO_4)_3.14H_2O$	594.4		Liquid	8.5 (Al_2O_3)
				Lump	17 (Al_2O_3)
Aluminum Chloride	$AlCl_3$	133.3	44	Liquid	-
Lime	$Ca(OH)_2$	56.1 (as CaO)	40	Lump	63-73 (as CAO)
				Powder	85-99
				Slury	15-20
Ferric Chloride	$FeCl_3$	162.2	91	Liquid	20 (Fe)
				Lump	20 (Fe)
Ferric Sulfate	$Fe_2(SO_4)_3$	400	51.5	Granular	18.5 (Fe)
Ferric Sulfate (copperas)	$FeSO_47H_2O$	278.1	139	Granular	20 (Fe)
Sodium aluminate	$Na_2Al_2O_4$	163.9	100	Flake	46 (Al_2O_3)

2.2.1 Application Points

Mineral salts could be added to the wastewater to precipitate phosphorus in several points such as before primary sedimentation, to secondary processes and simultaneous combination of these alternatives (multiple points) (WPCF, 1983; Metcalf & Eddy, 2003)

2.2.1.1 Mineral salt addition before primary sedimentation

One of the main advantages of mineral addition before primary sedimentation (PS) is availability of sufficient mixing and flocculation environment and reduction of BOD and SS load to the biological stages due to improved removal efficiency. The main disadvantage of this method is originated from the form the polyphosphorus compounds that could not be precipitated easily. In the raw wastewater, the percentage of phosphate that could be easily precipitated is lower as compared to polyphosphorus. Figure 2.2 illustrates the representative scheme of mineral addition before PS.

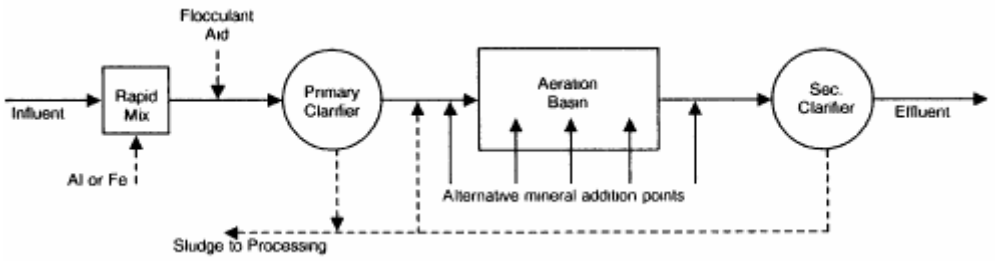


Figure 2.2 Mineral salt addition before PS

Properly designed and operated precipitation systems could achieve 70-90% P removal efficiency in addition to improved SS and BOD removal efficiency. Since the metal salts react not only with phosphate but also react with other ionic forms, metal salt demand is higher than the secondary treatment metal salt addition method (WPCF, 1983; USEPA, 1987).

2.2.1.2 Mineral salt addition to secondary processes

Addition of aluminum or iron salts directly to aeration basin is one of the common phosphorus precipitation methods. This alternative may provide with operational flexibility from the point of chemical addition, modification of the dosage point to ensure use of the best available conditions for coagulation and flocculation to occur. Figure 2.3 illustrates the representative scheme of mineral addition to the secondary processes including mineral addition before aeration basin, directly injection of metal salts into the aeration basin and before final clarifiers.

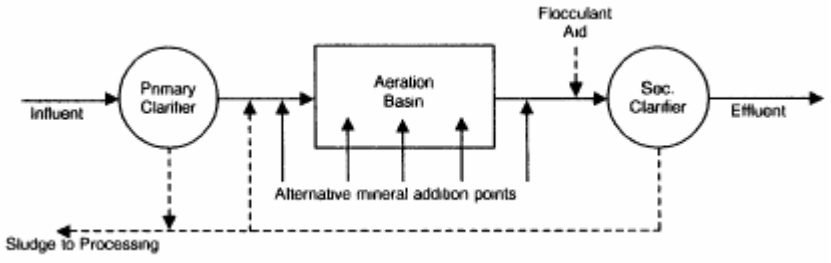


Figure 2.3 Mineral salt addition to the secondary processes

One of the most important disadvantages of this method is presence of insufficient velocity gradients or turbulence levels for precipitation reactions. Addition of mineral salts to the secondary treatment processes may result with an increase in the effluent dissolved solids due to impure chemical sources. When aluminum or iron salts are used for the phosphorus precipitation, addition of small amounts of an anionic polyelectrolyte (0.1-0.25 mg/L) should be necessary to remove some additional dispersed metal-phosphate floc.

2.2.1.3 Mineral salt addition at multiple points

Addition of mineral salts at multiple locations in the treatment plants has been found to be efficient and cost-effective methods. Figure 2.4 illustrates the representative scheme of mineral addition at multiple points. This alternative should be considered at design level of new facilities to create operational flexibility, optimize the chemical dosages and provide better phosphorus control performance.

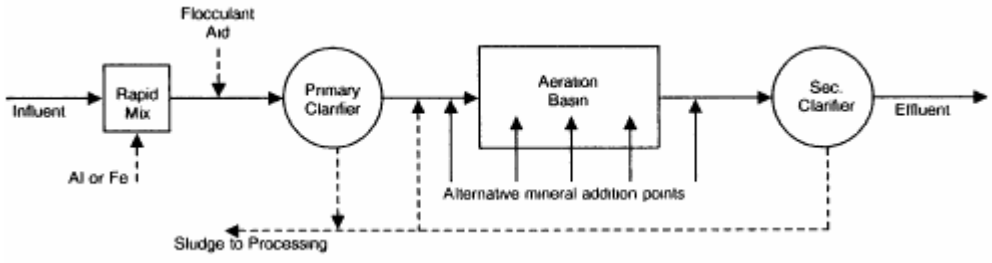


Figure 2.4 Mineral salt addition at multiple points

Table 2.4 Potential effectiveness of primary and secondary treatment with and without mineral addition (USEPA, 1987)

	Phosphorus Removal (%)		SS Removal (%)		BOD Removal (%)	
	Without	With	Without	With	Without	With
Primary Treatment	5-10	70-90	40-70	60-75	25-40	40-65
Activated Sludge	10-20	80-95	80-95	85-95	85-95	85-95

As it could be seen from the Table 2.4, properly designed chemical precipitation systems could increase the phosphorus, suspended solid (SS) and BOD removal efficiencies.

Table 2.5 Required chemical dosages according to the addition points addition.

Point of addition	Chemical	Average metal ion/TP ratio		
		WPCF, 1983	USEPA, 1987	Metcalf & Eddy, 2003
Raw Wastewater	Ferric chloride	2.7	2.7	2.3
	Alum	1.7	1.7	-
Mixed liquor	Ferric chloride	1.5	1.5	-
	Alum	1.6	1.6	-

Required chemical dosage to obtain effluent TP lower than 1 mg/L was investigated in approximately 50 full-scale wastewater treatment plants in the USA and the average of this study is given Table 2.5 As it could be seen the above given table, required metal ion is decreased in secondary treatment addition point.

2.2.2 Evaluation of the existing processes to retrofit for chemical precipitation

Retrofit applications for phosphorus precipitation within the treatment plant should be performed by considering availability of chemical addition points to guarantee the adequate mixing, providing with proper conditions for flocculation and adequacy of the existing clarifiers (Janssen, Meinema, van der Roest, 2002; Metcalf & Eddy, 2003). In addition to these factors, evaluation of impacts on aeration requirements, SRT, F/M and other key process parameters is critical as well. May be the most important concern should be given to the increase of sludge quantity as result of chemical precipitation and operational problems associated with unit operations for chemical sludge. The negative effects of the chemical sludge on sludge treatment facilities such as thickening, dewatering, thermal applications including drying and incineration should also be taken into consideration. One of the most important retrofit considerations is definitely increased sludge volume changed sludge characteristics in the plant. Increase of total sludge quantity could be explained with formation of precipitants such as metal phosphates and metal hydroxides; improved solids removal efficiency in final clarifiers; removal of dissolved solids (WPCF, 1983; USEPA, 1987).

2.2.3 Handling of chemical sludges

It was reported by USEPA (1987) that sludge characteristics of the sludge formed with aluminum salts is far a way different from sludge formed without chemical addition. While in some cases thickening/dewatering processes were positively affected, detrimental effects of chemical sludge on thickening/dewatering processes were also reported. In any case, many serious problems could be expected overloading effect due to increased sludge quantity. Regarding a typical WWTP sludge handling scheme, thickening process was followed by a stabilization method (mainly anaerobic digestion) and a final disposal method including thermal methods such as drying and incineration.

One of the main prerequisites of the sludge thickening and dewatering process is chemical conditioning. Many types of polyelectrolyte were used to improve the thickening and dewatering performances (Metcalf & Eddy, 2003). It could be safely expressed that the chemical conditioned requirement will increase as a result of increased sludge quantity and it was reported that 40% increase could be expected when alum is used as a precipitant. It was also reported that dewatering efficiency and sludge recovery rate could also adversely affected use of chemicals in phosphorus precipitation. It was also reported that dewatering of ferric chloride sludge was more difficult as compared to the both alum sludge and non-chemical sludge (USEPA, 1987).

A laboratory-scaled investigation indicated that anaerobic digestibility of chemical sludge precipitated with alum was lower than the non-chemical sludge. This finding explained with association of substrate within in coagulant floc, rendering some portion of the organics less accessible for the anaerobic bacteria. Another study will resulted with similar findings of decreased methane production rate, volatile solids reduction rate and COD utilization (WPCF, 1983). USEPA surveyed treatment plants in the USA to determine the anaerobic digestion of chemical sludge formed with alum. This study revealed several, significant detrimental effects of alum-precipitated sludge on anaerobic digestion process including; increased energy requirement for heating, pumping and mixing, difficulty in maintaining adequate mixing and heating, increased maintenance requirement for sludge pumping; poor solids-liquid separation and reduction in digester efficiency.

Incineration is one of the most applied methods for the final disposal of wastewater treatment sludge. Increasing public concern on environmental issues and new strict regulations made the incineration choice is one of the most popular methods. Incineration of sewage sludge includes several critical parameters. Moisture content of the sludge, calorific value of the sludge and relative portion of volatile solids to inorganic solids influence the incineration process significantly. Although any specific study was implemented, a significant increase in energy demand could be expected because of the increased sludge production rate and decreased sludge dewatering performance (increase of moisture content dewatered sludge).

2.3 Enhanced Biological Phosphorus Removal (EBPR)

2.3.1 Theory of EBPR

It has been proved by many experimental studies that exposing the mixed liquor to an anaerobic/aerobic sequence results with selection of microorganisms that able to store higher levels of intracellular phosphorus than other microorganisms (Park et al, 2001; Reddy, 1991; Bradjanovic et. al., 1998). Phosphorus-removing microorganisms are able to rapidly assimilate and store volatile fatty acids (VFAs) and other fermentation products under anaerobic conditions. Phosphorus is released to the anaerobic zone to produce the energy needed to take up the fermentation products, which are stored as poly-ß-hydroxybutyrate (PHB). Phosphorus-removing microorganisms produce energy by oxidizing the stored fermentation products in the aerobic zone while simultaneously accumulating intracellular phosphate. The ability of phosphorus-removing microorganisms to rapidly assimilate the fermentation products under anaerobic conditions gives them a competitive advantage over other microorganisms and results in their preferential growth in the activated sludge. Thus, the anaerobic-aerobic sequence allows the selection of a large population of phosphorus-removing microorganisms (Comeau et. al, 1986 ; Wentzel et. al., 1991).

Acetate and other fermentation products are produced from fermentation reactions by normally occurring facultative organisms in anaerobic zone. These fermentation products are derived from the soluble portion of the influent BOD and there is not sufficient time for the hydrolysis and conversion of the influent particulate BOD (Park et. al, 2001; USEPA, 1987). The fermentation products are preferred and readily assimilated and stored by the microorganisms capable of excess biological phosphorus removal. This assimilation and storage is aided by the energy made available form the hydrolysis of the stored polyphosphates during anaerobic period. The stored polyphosphate provides energy for active transport of substrate and for formation of acetoacetate, which is converted to PHB (Metcalf & Eddy, 2001; 2003; Comeau et. al, 1986 ; Wentzel et. al., 1991).

During aerobic phase, the stored substrate products are depleted and soluble phosphorus is taken up, with ***excess amounts*** stored as polyphosphates within volutin granules. An increase in the population of phosphorus storing bacteria is also expected as a result of substrate utilization. The above mechanism indicates that the level of biological phosphorus removal achieved is directly related to the amount of substrate that can be fermented by normally occurring microorganisms. In the anaerobic zone and subsequently assimilated and stored as fermentation products by phosphorus. removing microorganisms also in the anaerobic phase (Satoh et. al., 1996; Sudiana et. al., 1999).

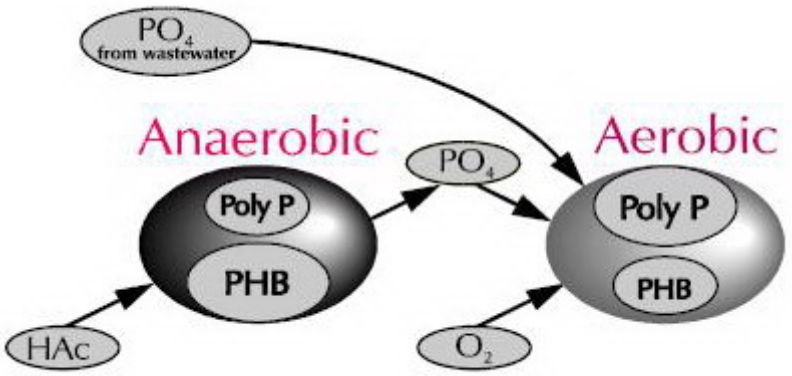


Figure 2.5 Theory for release and uptake of phosphorus by PAOs.

In addition to well-known theory of EBPR, anoxic phosphorus uptake by denitrifying bacteria was defined by many investigators. Concept of the theory is based on anaerobic phosphorus release and anoxic phosphorus uptake using nitrate instead of DO as electron acceptor (Kuba, 1994; Lee et al., 2003)

2.3.2 Pathways of EBPR

The most widely accepted of these models have been the Comeau/Wentzel model (based on Comeau *et al.*, 1986), the Mino model (Mino *et al.,* 1987) and the Adapted Mino model (Wentzel *et al.*, 1991). In addition, modifications of these models and new mechanisms have been proposed by Pereira *et al.* (1996); Louie *et al.* (2000); Maurer *et al.* (1997); and Sudiana *et al.* (1999). It was observed that studies explaining biochemical models of EEBPR differ from each other on generation of reducing power in anaerobic zone.

First conceptual biochemical model for EBPR was described by Comeau *et al.* (1986). They suggested that TCA cycle creates required reducing equivalents necessary to reduce acetyl-CoA to PHB. In their studies glycogen formation for storage purposes was not included explaining fermentative reactions would be more favorable compared to glycogen formation and that the bulk solution needed to contain a high concentration of sugars for the glycogen storage to occur, with or without PHA synthesis.

Different from the Comeau/Wentzel model, the Mino and Adapted Mino models predict that glycogen, an intracellular carbohydrate reserve, serves as a supplemental electron donor for PHA production. Thus, it is degraded in the anaerobic stage (Mino *et al.,* 1987 and Smolders *et al.,* 1994) for this purpose, and in the following aerobic stage, PHA is broken down as a carbon and energy source to: synthesize new cells, produce the reducing equivalents (NADH) needed for ATP production and restore the depleted glycogen reserves. The newly generated ATP is used by cells for energy or to store poly-phosphate granules, which are later used as an energy source for acetate uptake and PHA storage in the anaerobic zone. In the Comeau/Wentzel model, acetate or other VFAs are taken up by the cells and directed through the TCA cycle. In the simple case, a portion of the acetyl-CoA is directed to the TCA cycle and NADH generated from the cycle interacts with the remaining acetyl CoA to produce PHB. In the Mino Model, reducing power is generated through breakdown of stored glycogen, which is recycled from the aerobic stage. Glucose-1-phosphate is

then directed through the Embden-Meyerhoff-Parnas (EMP) pathway to create reducing power. The Adapted Mino model is similar to the Mino model in the general outlook of events, but the former one considers the Entner-Doudoroff (ED) pathway as the NADH source. Researchers agree on the storage of organics in the form of PHA polymer, which is a complex polymer of polyhydroxy-butyrate, polyhydroxyvalerate and their methylated forms (Satoh et al., 1992). PHA oxidation and intracellular phosphate storage in the aerobic stage also are defined in all models. However, the Mino and Modified Mino models additionally suggest glycogen formation in the aerobic stage.

A new model called "Adapted Mino Model" was suggested by Wentzel et al. (1991) that were prepared evaluating the existing biochemical models. This new model was proposing that the glucose degradation following glycogen breakdown proceeded through the Entner-Doudoroff (ED) pathway, rather than the Embden-Meyerhoff-Parnas (EMP) pathway as originally suggested by Mino and his co-workers.

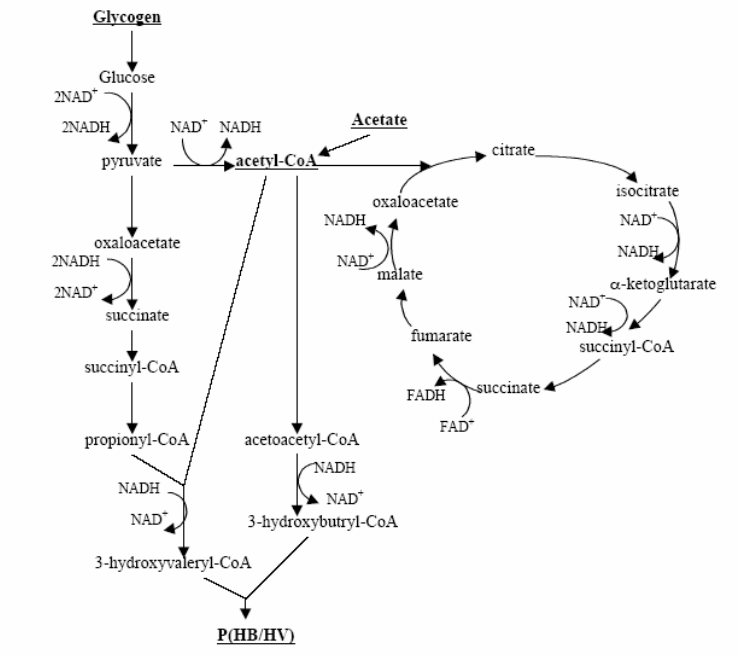


Figure 2.6 EEBPR model developed by Pereira and her co-workers (1996)

2.3.3 Parameters Effecting EBPR

2.3.3.1 Wastewater characteristics

Accurate characterization of wastewater as concentration and loading rates plays an important role in EBPR. Physical characteristics and chemical composition of wastewater should be well defined as nutrients and their various forms. According to the well known theory of EBPR, influent carbon source level directly controls effluent quality in many cases (Metcalf & Eddy, 2003; Park et al., 2001). Especially; availability of rbsCOD in anaerobic zone is one of the essential considerations. It was reported that at least 20 mg as acetic acid (Janssen, Meinema, van der Roest, 2002; Abu-ghararah, 1991), 50 mg as COD (Ekama and Marais, 1984) and 7 – 9 mg as VFAs (Barnard, 1993) are required to remove 1 mg of P. According to the scientific investigations, EBPR process could be considered as COD limited when the COD/TP ratio is low (<<20:1 for settled domestic sewage), whereas it is P limited when the COD/TP ratio is high. While low COD/TP ratios could cause EBPR failures, very low effluent P concentrations achievable at sufficient COD/TP ratios. It was reported that COD/TP ratio 40:1 are required to achieve effluent total P concentration 1 mg/l or less (Randall et al., 1992). Another study showed that BOD:P ratio should be at least 15-20 and BOD:N ratio 4-5 to guarantee EBPR efficiency in case of negligible electron acceptors input to the anaerobic zone (Janssen, Meinema, van der Roest, 2002).

Establishing a successful EBPR process mainly depends on accurate characterization of wastewater as both concentration and loading rates. Hydraulic and organic loading rates should be equalized and over loadings to the biological stages should be prevented (Shehab, Deininger, Porta & Wojewski, 1996).

The ratio of influent BOD or COD/P is another critical parameter in EBPR systems. To maintain a stable process, C.W. Randall, Barnard & Stensel (1992), proposed that BOD_5/TP ratio 20:1 and COD/TP ratio 40:1 are required to achieve effluent total phosphorus concentration 1 mg/l or less.

In spite the fact that existence of VFAs is vital for maintaining a stable EBPR process, excess amounts of VFAs in influent could deteriorate EBPR efficiency as well. Randall & Chapin, (1995) reported that acetate concentration above 600 mg/L in influent caused cessation of phosphorus release leading deterioration of removal efficiency.

Type of carbon source utilized in anaerobic zone is another important consideration in phosphorus removal. In fact, many investigators found a linear correlation between anaerobic COD utilization, phosphorus release and uptake. Abu-ghararah & Randall (1991) reported that phosphorus uptake/release rate was 1.2 with a correlation coefficient of 0.99. Similar results were obtained by Park, Whang, L.M., Wang, J. & Novotny (2001). They showed that the ratio was between 1.15 and 1.2. Satoh et. al., (1996) studied effect of different carbon sources on anaerobic phosphorus release and substrate consumption rates. They reported that maximum anaerobic phosphorus release was achieved using acetate and propionate and decreasing rates were observed with lactate, succinate, malate and pyruvate.

2.3.3.2 Effects of temperature and solids retention time

In addition to influent composition, environmental factors such as pH, temperature etc. effects EBPR process. Investigations proposed that substrate consumption and P release rates adversely affected by acidic pH in anaerobic zone; in addition, alkaline pH levels resulted with inhibition of acetate uptake but stimulated P release (Janssen, Meinema, van der Roest, 2002; Liu et al., 1996; Converti et al, 1995).

Temperature is another critical parameter for EBPR similar to all biochemical reactions. Generally P release and P uptake rates increase with increasing process temperature (Janssen, Meinema, van der Roest, 2002). Both short and long term investigation results indicated that temperature had a rather strong impact on anaerobic metabolism kinetics. In contrast to anaerobic phase a uniform temperature dependency of metabolic processes of the aerobic phase was not observed. It was

also reported that temperature strongly affected oxygen and poly hydroxyl alkanoate (PHA) consumption rates. Another important effect of temperature on activated sludge was changing composition of the microbial culture at different temperatures (Brdjanovic et al., 1997).

Temperature has significant effect on all biological treatment processes. Efficiency of a biological treatment system is directly affected by temperature shifts as result of changing metabolic activity of the microbial culture and settling characteristics of the sludge determining directly effluent quality are a function of the temperature.

Erdal, Z.K. Erdal, Randal (2003) explored the metabolism of phosphorus removal under temperature controlled conditions. EEBPR sludges were cultivated in two separate lab-scale UCT system operated at 5 ˚C and 20 ˚C. After an adequate acclimation period, at both temperatures, system functions were successful. They found that phosphorus removal performance was optimum at SRT ranges of 16-24 days and 12 to 17 days for 5 and 10 ˚C. Higher SRT values up to 32 days at 5 ˚C and 25 days at 10 ˚C reduced EEBPR performance. This deterioration was explained by increased extent of endogenous respiration which consumed internally stored glycogen, leaving less reducing power of PHA formation in anaerobic stages. The washout SRT of each system found as 3.5 days at 5 ˚C and 1.8 days 5 ˚C. They concluded that EEBPR microorganisms are capable to alter their biochemical metabolism based on environmental conditions.

Shell, (1981) observed that phosphorus removal efficiency at 5 ˚C was greater by more than %40, ef at 15 ˚C and this situation was explained by investigators that microorganism culture shift to slow growing bacteria with a higher cell yield.

Brdjanovic, van Loosdrecht, Hooijmans, Alaerts, Heijnen, (1997) reported that temperature affects oxygen utilization rate significantly in combined EBPR systems. While P uptake rate at 5 ˚C and 10 ˚C was insufficient, at 20 ˚C and 30 ˚C complete p uptake was observed.

Both short and long term investigation results indicated that temperature had a rather strong impact on anaerobic metabolism kinetics. In contrast to anaerobic phase a uniform temperature dependency of metabolic processes of the aerobic phase was not observed. It was also reported that temperature strongly affected oxygen and poly hydroxyl alkanoate (PHA) consumption rates. Another important effect of temperature on activated sludge was changing composition of the microbial culture at different temperatures (Brdjanovic et al., 1997).

McClintock, Randall & Pattarkine (1992), observed that at a temperature of 10 ˚C and SRT of 5 days, Enhanced Biological Phosphorus Removal System (EEBPR) would "wash-out" before other heterotrophic functions do. Mamais & Jenkins (1992) showed that there is a wash – out SRT for all temperatures over the range of 10 to 30 ˚C. Their investigations have showed that if the temperature and SRT combination is lower than a limit value, EEBPR system performance stops although EEBPR performance increases at lower temperature.

Although revised reports include conflicts on temperature effect, in combined BNR systems, nitrification and denitrification process are deteriorated with decreasing temperatures. (Helmer & Knust 1998; Wagner, Noguera, Juretschko, Rath, Knoops, Schleifer (1998). High nitrate concentrations in return sludge to anaerobic zone of EBPR systems may occur in lower temperatures that ends with consumption of available substrates for PAOs by denitrifiers resulting with poor phosphorus removal efficiency. Another adverse effect of low temperature causes from selection of filamentous bacteria such as *Microthrix parvicella* that has an optimum growth temperature of ≤15-12 and that creates bulking sludge problem. (Knoop & Kunst, 1998). It could be concluded that decreasing temperature has adverse effect on EBPR efficiency.

2.3.3.3 Effect of pH

Schuler & Jenkins (2002) reported higher acetate uptake rate by a PAO dominated system at a pH of around 7 or greater, while a mixed PAO/GAO system had higher

uptake rates as the pH dropped below about 6.8. Experiments performed at pH 7.15-7.25 where PAOs were clearly dominant have also been reported, though (Schuler and Jenkins, 2002).

McGrath, Cleary, Mullan & Quinn (2001) examined acid – stimulated phosphorus uptake by activated sludge obtained from five different wastewater treatment plants. Microorganisms were grown aerobically under laboratory conditions on mineral salts medium containing either glucose or skimmed milk powder as carbon source. More than 50% and 143% phosphorus uptake at growth pH 5,5 were achieved without sequencing the microorganisms to anaerobic aerobic conditions. After 24 h growth at pH 7,5 the inoculums removed 0,39 mmol/l phosphate from the growth media. The microorganisms removed 0,585 mmol/l phosphate (50% enhancement) growth at pH 5,5 during stationary phase. They concluded that optimum pH range for phosphorus uptake was between 5.5 and 6.5.

Liu, Mino, Matsuo & Nakamura (1996) reported that substrate consumption and phosphorus release rates adversely affected by acidic pH in anaerobic zone; in addition, alkaline pH levels resulted with inhibition of acetate uptake but stimulated phosphorus release. Similar results were obtained by Converti, Rovatti & Del Borghi, (1995). Decrease in pH from 7.2 to 6.3 resulted with efficiency decrease in the EBPR system.

It is clear that to maintain a stable biological treatment efficiency, monitoring the pH value in different zones of EBPR such as anaerobic, anoxic and aerobic is vital and keeping the activated sludge mixture at neutral pH is one of the key parameter of the successful EBPR systems.

2.3.3.4 Presence of electron acceptors in anaerobic zone

Concentration of DO or nitrates in the anaerobic zone should be minimized in order to obtain high removal efficiencies of nutrients. These electron acceptors were mainly carried to anaerobic zone by influent and recycle stream in the form of both

nitrate and dissolved oxygen. Organic matter that is required for poly-P bacteria will be oxidized by the ordinary heterotrophic bacteria in the presence of electron acceptors. The reduction of EBPR efficiency was based on decrease of ORP (Oxygen Reduction Potential) due to existence of electron acceptors in anaerobic zone. Similar results of reported by USEPA (1987) that total phosphorus removal efficiency decreased from 90 to 55 when effluent nitrate concentration increased from 4.0 to 6.7 mg/L. Another study that was cited by USEPA (1987), conducted on wastewater with high BOD/P ratio, showed that although effluent nitrate concentrations as high as 6.7 to 11.6 mg/L, effluent phosphorus concentrations were lower than 1 mg/L. These two different results could be concluded that effect of electron acceptors on EBPR efficiency depends mainly on wastewater BOD/P ratio. USEPA (1987) reported that phosphorus release in anaerobic phase was inversely proportional to the amount of nitrogen present when excess substrate available. Denitrification of nitrate in anaerobic zone had the effect of reducing the availability of substrate for phosphorus release.

It was reported that presence of excess DO was causing poor performance at a number of full-scale EBPR systems in South Africa. It was also reported that in combination with weak wastewater composition, high DO concentration in influent was susceptible of causing poor phosphorus removal and growth of filamentous bacteria in some EBPR facilities (USEPA, 1987).

Recent molecular identification techniques such as fluorescence in situ hybridization analysis provided better understanding for EBPR mechanism. Fish analysis combined with laser scanning microscopy of EBPR system showed that change of the electron acceptors from oxygen to nitrate resulted with a population shift in bacteria population from alpha subclass to filamentous, beta subclass bacteria with in two weeks (Falkentoft et. al. 2002).

2.3.4 Responsible Microorganisms in EBPR

Lötter & Murphy (1985), studied on the identification of heterotrophic bacteria in an activated sludge plant with particular reference to polyphosphate accumulation. They found that *Pseudomonas*, *Aeromonas* and *Acinetobacter* accomplished denitrification in the anoxic stage of a biological nitrogen removal system. Jorgensen and Pauli (1995) confirmed that the denitrifiers, such as *Pseudomonas, Hydrogenophaga, Citrobacter, Xanthomonas*, have the ability of phosphorus accumulation. The excess phosphorus uptake rates of these bacteria were higher than those reported for *Acinetobacter strains*. These results showed that polyphosphate accumulation and denitrification in activated sludge can be carried out by the same microorganisms. It has been reported that *Pseudomonas* and *Acinetobacter* in an anoxic stage coexisted at 55% and 16%, respectively. Also, it has been suggested *Pseudomonas* are actively take part in simultaneous denitrification and phosphorus uptake in EEBPR sludge (Atkinson *et al*., 2001).

Lee S. H. et. al., (2003) studied on verification of the microbial community in a EEBPR system, the identification of microbiology was accomplished using the PCR-DGGE method. The species was detected using DNA extraction, PCR Amplification and DGGE (Dcode Universal Mutation Detection System (BioRad, USA) methods and found two genera: *Dechloromonas* and *Rhodocyclus*. However, *Acinetobacter sp.*, *Aeromonas sp*. and *Pseudomonas sp*., which historically have been considered dominant in phosphorus removing sludge communities, were not detected.

2.4 Improvement of EBPR process by primary sludge fermentation (PSF)

Prefermentation is a new and increasingly popular unit operation in BNR treatment systems with the role of producing VFAs, which could enhance EEBPR systems. Weak wastewater characteristics, especially low readily soluble biodegradable substances, may deteriorate desired treatment levels in BNR plants. Sewage systems with short retention time and operation at low temperatures could be possible reasons of insufficient VFAs level in influent. In these cases additional

soluble organic material is required which could be obtained by either prefermentation of both primary sludge or wastewater it self or purchasing carbon source from markets. Many investigation results showed that production of VFAs by prefermentation could be a cost effective method (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

2.4.1 Mechanism of prefermentation

Fermentation involves two initial steps of anaerobic digestion. Under anaerobic conditions, particulate or complex soluble organic substrates are broken down through extra cellular enzymatic hydrolysis reactions into VFAs and methane by anaerobic microorganisms. Fermentation reactions includes to main steps;

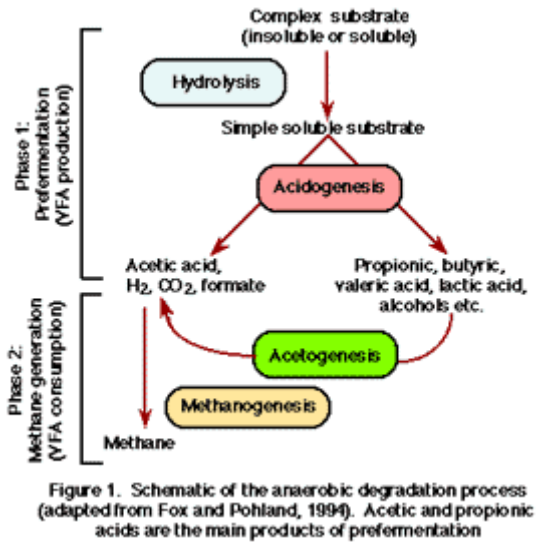
Hydrolysis: Particulate or high molecular weight soluble substrates are broken down to simpler forms incorporation with water molecules and catalyzed by hydrolytic enzymes excreted by the bacteria. (Sugars, fatty acids, and amino acids.) **Acetogenesis**: Fermentation of organic monomers to short chain organic compounds such as alcohols, acetate, butyrate, and propionate.	 Figure 1. Schematic of the anaerobic degradation process (adapted from Fox and Pohland, 1994). Acetic and propionic acids are the main products of prefermentation

Figure 2.7 Mechanism of acidogenic fermentation

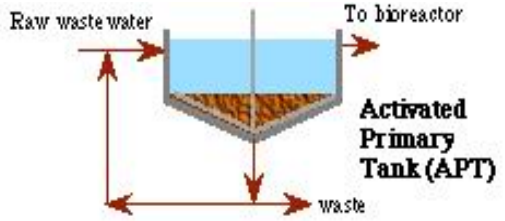
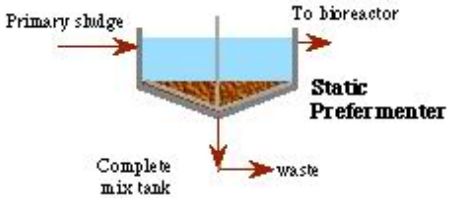
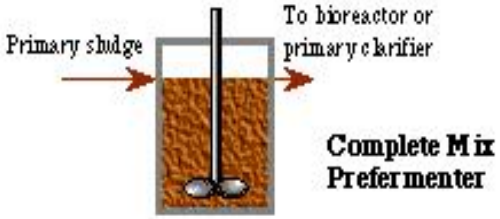
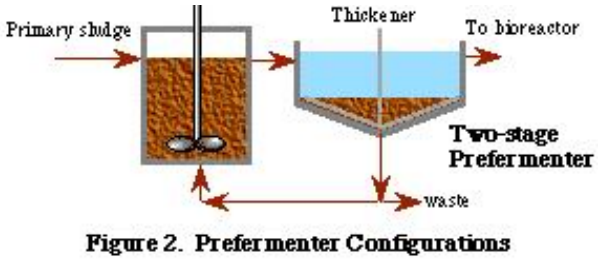
Activated Primary Tanks (APT): APT produce VFAs allowing a sludge blanket in primary clarifier. The sludge is recycled to the inlet to inoculate Suspended solids with actively fermenting microorganisms.	
Single-Stage, Non-mixed (Static) Prefermenter: The Solids Retention Time controlled by keeping the sludge blanket height constant.	
Complete Mix (Single Stage) Prefermenter: Fermented sludge is fed to bio reactor or primary clarifier.	
Two-Stage Complete Mix. Prefermenter: After a complete mix. Reactor VFAs are elutriated in a clarifier or dedicated thickener.	

Figure 2.8 Prefermenter configurations

2.4.2 Prefermenter design & operation fundamentals

2.4.2.1 Specific VFAs generation rate

These parameters describe performance of a thickener and they can be controlled separately depending on fermenter design. Elutriation (wash out) of VFAs has more importance on the VFA recovery than specific generation rate. The most practical way to measure the VFA yield is to relate it to the volatile suspended solids fed to the fermenter and could be expressed as follows;

$$Specific VFAs generation = \left[\text{VFA generated (kg/d)} \Big/ \text{VSS fed to fermenter (kg/d)} \right] \times 100$$

2.4.2.2 Temperature and oxygen reduction potential (ORP)

Metabolic activities of fermenting microorganisms increase with temperature. Maximum VFA production yield could be achieved at higher temperatures at lower SRT values. At the same time methane formation also increases with the increasing temperatures so VFA yield may decrease at higher temperatures as a result of methane formation. (Baur, Bhattararai, Benisch, Neetling, 2002). Randall et. al., (2002), stated that acidogenic fermentation occurs even above -300 mV, while methanogenic fermentation occurs below -500 mV. Barajas, Escalas, Mujeriego (2001), reported that fermentation efficiency was deteriorated temperatures below 18 ˚C and cold influent conditions increased ORP from -460 mV to -375 mV. They reported that operation at 10 d SRT reduced ORP form -450 mV to -550 mV and shifted the reactions from acidogenic fermentation to methanogenic fermentation and VFA yields fell down to negative values due to methane formation.

2.4.2.3 Mixing

There are several opposite opinions on the effect of mixing intensity on VFA generation yield. Baur, Bhattararai, Benisch, Neetling, (2002), showed that mixing increases the fermenter yield by maintaining homogenous conditions within the

reactor and also provides better contact between substrates and microorganisms. Their full scale studies showed that while performance of complete mix fermenter is about 90%, it is between 20-50% for static fermenter.

Banister & Pretorius (1996), reported that VFA generation yield increased from 0.04 to 0.07 (for ORAW, USA wastewater with 3.8 %TS) and from 0.09 to 0.15 mg VFA (as COD)/mg COD (initial) in NRAW wastewater with 1.2 % TS in the mixed and unmixed fermenter. They have not recommended intensive mixing of fermenter inventory. They also reported that keeping the solids in suspension may also disturb the microenvironment in fermenter or allow for oxygen entrapment, inhibiting fermentation or allowing some of the VFA to be metabolized.

2.4.2.4 Elutriation of generated VFAs

The VFAs produced in the fermenter must be washed out of the biomass. Natural diffusion driven elutriation takes place due to VFAs concentration gradient available in a static fermenter. Increasing mixing intensity also increases the VFAs elutriation rates and thereby complete mix reactor outperforms VFAs generation than static mixers do.

Full scale fermentation studies conducted in WWTF by Baur, Bhattararai, Benisch, Neetling, (2002) were divided into two modes as series and parallel operation. In the parallel mode, both fermenters/thickeners were operated separately at different SRT. Elutriation was achieved by normal turbulence and diffusion of VFA from the sludge blanket into the supernatant. In the series operation mode, the firs reactor was operated as fermenter and following reactor was operated as a separator. The separator received the blended underflow and overflow from the fermenter. During the parallel mode, temperature ranged from 19.3 to 22.1 ^{o}C and SRT was kept between 1.6 and 5.5 days. Specific VFA generation rate (VFA/VS) were in the range of 5.52 and 11.35% and elutriation performance (%) was between15.5 and 59.9 %. The best result was obtained in this mode at temperature of 21.7 ^{o}C and SRT of 5.5 days as 11.35 specific VFA generation rate and elutriation

efficiency of 59.9 %. However closer performance was also obtained at temperature of 22.1 ^{o}C and SRT of *2.0 days* as 8.81 specific VFA generation rate and elutriation efficiency of 58.5 % were achieved. Series operation studies conducted at 18.8 ^{o}C and "1.8+1.0" days SRT resulted in 8.10 % specific VFA generation rate and 43.4 % elutriation efficiency.

2.4.2.5 Effect of seeding and dilution

Banister & Pretorius (1996) found that mixing of partially fermented sludge with fresh primary sludge boosted VFA generation yield. Shorter hydraulic retention times could be applied by seeding so smaller reactor volumes could be considered in design level. However size of inoculums and predominance of acidogens in the seed have significant importance on the system performance. An increase of seeding ratio from 10% to 20% resulted with no improvement in VFA yields.

Barajas, Escalas, Mujeriego (2001), investigated prefermentation of low volatile fatty acid wastewater in a laboratory – scale primary clarifier operated as a prefermenter – an activated primary tank. The prefermenter performance was studied over three operational periods. System was operated at 5 days, 10 days and 5 days (closed system) solids retention time. Wastewater flow rate was 2,5 l/h for the whole study. Recycle flow was 1.3 l/h (52%). Hydraulic retention time was 1,3 hours. During the all study temperature was in the range of 18,9 – 20,3 ^{o}C and oxygen reduction potential (ORP) shifted between -110 to -465 mV.

The best results were obtained in the covered system at 5 days SRT. Under these conditions, both COD solubilisation was 22 mg/L and VFA production was 34 mgVFA/l. The VFA / PO_4-P ratio was improved from 0,9 to 5,5 mg VFA – COD/mg PO_4-P but recommended ratio to obtain favorable conditions biological phosphorus removal (20 mg/mg) was not sustained.

Katehis et. al., (2003), studied use of fermented primary sludge as a BNR supplemental carbon source. Their study had two main objectives; the quality and the

quantity of fermentate and the effectiveness of the fermented sludge supernatant as a supplemental carbon source. Ambient fermentation were tested at two SRT/volatile solids loading rates; a low rate of 8-9 days SRT at 100-200 kg VSS/d and an increased rate of 4-4,5 days SRT at 300-600 kg VSS/d were examined. The main objective of the research was to supply additional carbon source for New York Water pollution Control plants (WPCP) receiving weak wastewater composition. While the average COD solubilisation during low rate operation was 1230 mg/l, it was 1760 mg/l for increased rate operation period. Total suspended solids concentration in fermenter decreased from 400-700 mg/l TSS to 300-400 mg/l TSS when operation was shifted from low rate to the increased rate. Ratio of sCOD to NH_3 (ammonia solubilisation) was between 20-50 during the low rate operation mode and 40-90 in the increased rate operation. The analytical measurements showed that % 85-90 of fermentate was including 1000-2000 mg/l VFA as in the form of mainly acetic and propionic acids. They reported that there is no need to a microorganism acclimation period to use of fermentate as a carbon source. Denitrification rates measured during the research were ranging from 0,12 to 0,28 mgN/mgVSS*d at 13-24 ˚C. Increased operation temperatures enhanced the fermentation efficiency by 30%. The VFA produced in the fermenter was highly suitable as a supplemental carbon source despite having a pH of 5,5 no buffering was required. The high kinetics was achieved from fermentate enabled denitrification to occur in smaller anoxic volumes and could decrease the sensitivity of the BNR plant. They found that soluble COD:NH_3 ratio was a function of SRT. They concluded that retrofitting a facility for primary sludge fermentation, including odor control could be a cost effective alternative to purchasing supplemental carbon source

2.4.2.6 Associated problems with PSF

Since some part of the organic material is acidified in fermentation reactions the quantity of the methane gas from anaerobic digestion process decreases. During the process high concentrations of odor compounds could be released. Therefore adequate odor control system is required. One of the main concerns is stimulated

filamentous bacteria growth as a result of long chain fatty acids fed from PSF process (Janssen, Meinema, van der Roest, 2002).

2.5 Comparison of EBPR and CPR

One of the most important advantages of EBPR over CPR is occurrence of no chemical sludge. CPR leads an increase in total sludge production because of improved suspended and dissolved solids removal efficiency, formation of metal – hydroxide precipitants etc. Increased sludge production rate also leads to higher costs of sludge treatment, disposal and final management (Janssen, Meinema, van der Roest, 2002).

It was also reported that chemical phosphorus precipitation is a very critical from biochemical point of view. When the dosage is too high the orthophosphate will be bound chemically and will not be available for PAOs for energy conversation. As result of that PAOs could not utilize simple substrate forms in the anaerobic period and finally, they could be washed out of the activated sludge system. Insufficient selection of PAOs will definitely results an increase of effluent phosphate concentration that will then require higher amount of metal salt dosage. In addition to increased sludge volume, dewaterability characteristics of the sludge could be deteriorated as a result of chemical phosphorus removal methods. In contrary to chemical methods, phosphate is preserved as potassium or magnesium phosphate (polyphosphate) without adhered water, resulting in a good dewaterable product (Janssen, Meinema, van der Roest, 2002).

Use of metal salts and lime for phosphorus removal not only negatively affects sludge dewatering characteristics but also influence other sludge disposal unit operations including anaerobic sludge digestibility, thermal drying and incineration of the sludge (USEPA, 1987). Since chemically removed sludge contains more adhered water, moisture content is higher than the non-chemical sludge. Therefore in combination of increased sludge volume and higher sludge moisture content, energy requirement that is one of the most determinative factors in operational costs will be

significantly higher as compared to non-chemical sludges from the point of sludge mixing, pumping, and heating (anaerobic digestion, thermal drying and incineration).

Another disadvantage of CPR is increased salinity and conductivity level in effluent originating from remaining negative ions (chloride and sulfide). In addition to increased salinity, effluent total dissolved solids (TDS) concentration could be increased as well due to impurities present in the metal salts (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

Addition of metal salts could destroy the wastewater buffer capacity that could result with inhibited or poorer nitrification efficiency especially for wastewater with week buffer capacity). Dosing of chemicals will remove the available COD not only for PAOs but also for the denitrifiers that means lowered denitrification efficiency. It could be concluded that CPR processes could deteriorate overall nitrogen removal efficiency (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

Phosphorus is one of the non-renewable and limited natural sources. Moreover, increased industrial and agricultural phosphorus demand increased the market price of the phosphorus (Isherwood, 2000). Investigations showed that recovery of chemically precipitated phosphorus is not possible or at least not feasible (Donnert and Salecker, 1999a; Donnert and Salecker, 1999b). In contrary, recovery of phosphorus from wastewater with an efficiency range of (10% -80%) is only possible with EBPR processes in which sludge became rich in phosphate (Strickland, 1999; Gaterell et al., 2000; Jeanmaire and Evans, 2001). It was also reported that fertilizer effect of EBPR sludge is attractive resource for agricultural purpose (Janssen, Meinema, van der Roest, 2002)

As it could be seen above given literature survey use of biological methods over chemical methods for removal of phosphorus from wastewaters have many advantages. However many scientific investigations were also demonstrated that EBPR removal processes are highly dependent upon influent wastewater characteristics such as; COD/TP ratio, pH and temperature etc. In addition, sludge

handling systems should be carefully designed and well operated to prevent phosphorus releases back to the plant (Meinema, van der Roest, 2002; Metcalf & Eddy, 2003). Therefore the stability of the EBPR process should be supported by CPR process to provide effluent discharge standards. However these systems should include fully automatic control systems

2.6 Biological nitrification

Nitrification is defined as a two-step biological reaction in which ammonia (NH_4-N) is oxidized to nitrite (NO_2-N) and then nitrite is oxidized to nitrate (NO_3-N). Nitrification process is performed by aerobic autotrophic bacteria in activated sludge process. The two groups of microorganisms responsible for the biological oxidation of nitrogen compounds are Nitrosomonas and Nitrobacter. Nitrosomonas oxidizes ammonia to nitrite. Nitrobacter completes the nitrification process by oxidizing nitrite to nitrate (Metcalf & Eddy, 2003).

The energy – yielding two-step oxidation of ammonia to nitrate is as follows:

1 st Step reactions:

$$2NH_4^+ + 3O_2 \xrightarrow{Nitrosomonas} 2NO_2^- + 4H^+ + H_2O$$

2 nd Step reactions:

$$2NO_2^- + O_2 \xrightarrow{Nitrobacter} 2NO_3^-$$

Nitrosomonas and Nitrobacter use the energy derived from these reactions for cell growth and maintenance. Total oxidation reaction is represented as follows:

Overall reactions:

$$NH_4^+ + 2O_2 \longrightarrow NO_3^- + 2H^+ + H_2O$$

In these biochemical reactions *Nitrosomonas* obtains more energy than *Nitrobacter*. Yield values calculated from theoretical energy release relations are 0.29g VSS/g of NH_4-N and 0.084 g VSS/g of NO_2^- (McCarty, 1964). According to the experimental studies, yield coefficients for oxidation of ammonium to nitrite by

Nitrosomonas, they are 0.04-0.13g VSS/g NH_4-N and for the oxidation of nitrite to nitrate by *Nitrobacter* 0.02-0.07g VSS/g NO_2-N (Painter, 1970). The total yield of nitrifies, when considering nitrification as a single-step process from ammonium to nitrate, is 0.06-0.20g VSS/g NH_4-N oxidized.

Nitrification process is sensitive to several environmental factors including temperature, pH, DO concentration and SRT.. The rate of nitrification in an activated sludge system decreases with decreasing temperature. Typical design SRT values may range from 10 to 20 d at 10 °C and 4 to 7 days for 20 °C. The optimum temperature is between 25 and 35 °C. The optimum pH for nitrification is in the range of 7.5-8.0 (Metcalf & Eddy, 2003).

Since alkalinity is destroyed during the nitrification process, pH could be expected to decrease. A ratio of 7.14 mg alkalinity is destroyed per mg of ammonia nitrogen oxidized. Aeration partially strips the carbon dioxide from the wastewater thereby reducing alkalinity reduction; however, sufficient alkalinity must remain in the wastewater so as not to depress the pH. Maximum nitrification rates occur at dissolved oxygen concentrations greater than 2 mg/L. For a complete nitrification process 4.57 g of O_2 is required per g of ammonium converted to nitrate (USEPA, 1993).

The fraction of nitrifying bacteria present in the activated sludge culture directly affects nitrification rate. A principal method of increasing the nitrification rate is to increase the fraction of nitrifiers. This could be achieved increasing the mixed liquor suspended solids (MLSS) concentration (USEPA, 1993).

2.7 Biological denitrification

The biological conversion of nitrate to nitric oxide, nitrous oxide and nitrogen gas is termed denitrification. In biological processes nitrogen is removed by assimilating and dissimilating nitrate reduction. In assimilating reactions nitrate is reduced to ammonia in cell synthesis. In dissimilating nitrate reduction or denitrification, nitrite

or nitrate is used as electron acceptor for the oxidation of various organic compounds (Metcalf & Eddy, 2003).

Unlike nitrification, a wide range of bacteria has been proved capable of denitrification. Denitrifiers are ubiquitous in most natural environments, including municipal wastewater and sludge (Christensen & Harremoes, 1977 ; Tiedje et al., 1982). Many of the microorganisms in municipal activated sludge systems are denitrifiers, even in systems that are not specifically designed for denitrifying. The domination of the denitrifiers in activated sludge culture is attributed to their ability to use either oxygen or nitrate as their terminal electron acceptor. Most of these bacteria facultative aerobic organisms with the ability to use oxygen or nitrate; nitrate and some could carry on fermentation in anaerobic environment (Metcalf & Eddy, 2003).

Environmental factors including temperature, pH, and dissolved oxygen concentration have an effect on the rate of denitrification. Denitrification occurs at temperatures in the range of 10-30 ˚C. The rate of denitrification is optimum at pH 6.0 to 8.0. Dissolved oxygen may inhibit denitrification by repressing the nitrate reduction enzyme. Dissolved oxygen concentration above 0.2 mg/l may inhibit biological denitrification. During nitrate reduction, 3.57 g of alkalinity as ($CaCO_3$) is produced per g of utilized nitrate (Metcalf & Eddy, 2003).

The stoichiometry of denitrification reactions could be expressed as follows:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

In denitrification reactions, the source of electron donor could be:

- Soluble readily biodegradable COD Influent wastewater
- Soluble readily biodegradable COD released during endogenous decay
- An external source including methanol and acetate

2.8 Historical Development of EBPR Treatment Plants

Barnard developed the Bardenpho process in 1974. This process was including separate anoxic and aerobic tanks. The settled sludge was recycled to the anoxic reactor, and providing an additional recycle from the aerobic to the anoxic reactor for the return of nitrates. By this configuration, TN concentration of effluent decreased by additional denitrification. The basic process configuration used in EBPR consists of an anaerobic zone followed by an aerobic zone. Barnard (1974) was the first researcher to find out the need for an anaerobic contact between organic matter in the influent and activated sludge and as a result of this knowledge the Bardenpho process was developed in 1974 by Barnard. The A/O process is a patented version of Barnard's findings by Air Products and Chemicals, Inc, and the main difference is the use of multiple-staged anaerobic and aerobic reactors. In these processes there is no nitrification, and the anaerobic retention time is 30 min to 1 h to provide the selective condition for biological phosphorus removal. The design aerobic SRT of these configurations varied from 2 to 4 d, depending on the temperature. A schematic view of Phoredox (A/O) configuration was depicted in Figure 2.9.

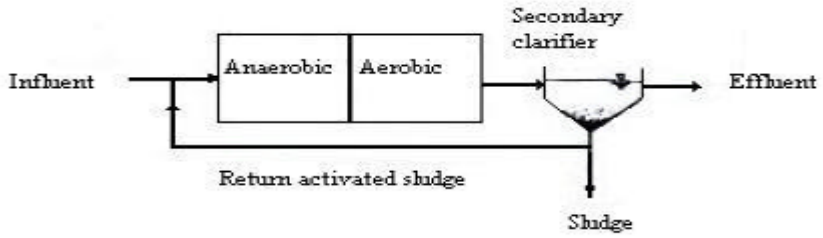


Figure 2.9 Flow diagram of Phoredox (A/O)

A process configuration for the utilization of EBPR was first reported by Levin and Shapiro (1965). It was proved that exposing the activated sludge to anaerobic/aerobic conditions resulted with selection of bacteria that could store excess amounts of P. However, the linkage between organic substrate storage and phosphorus release was not understood and the adverse effects of electron acceptors in the anaerobic zone was unknown. As a result of developments, a side stream process named PhoStrip, was used to accomplish P removal. This process combined biological and chemical removal of P.

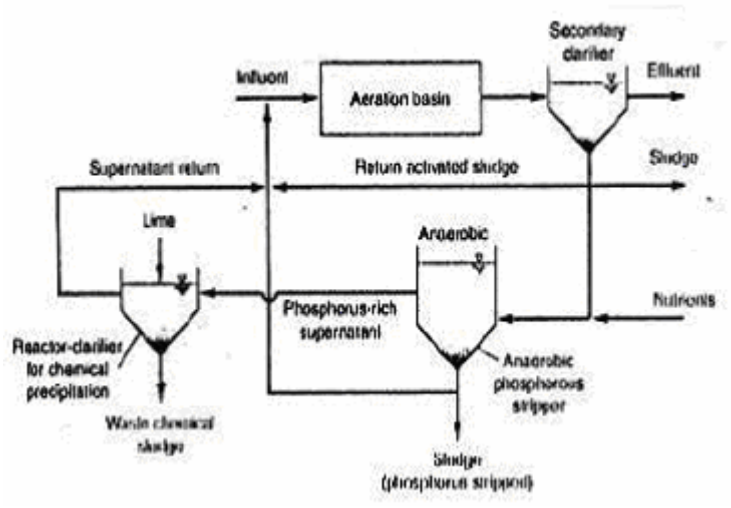


Figure 2.10 Flow diagram of Phoredox (A/O)

After pilot-plant and full-scale experimentation, Barnard combined the Phoredox concept with the MLE process at Klerksdorp, South Africa, in 1974. This resulted in the three-stage process called, and trademarked as "Anaerobic/ Anoxic/ Oxic", or A^2/O process which was later patented in the United States by Air Products and Chemicals, Inc. The proprietary A^2 / O process is a modification of the A / O process and provides an anoxic zone for denitrification. The detention period in the anoxic zone is approximately 1 h. The anoxic zone is deficient in dissolved oxygen, but chemically bound oxygen in the form of nitrate or nitrite is introduced by recycling nitrate from the aerobic zone.

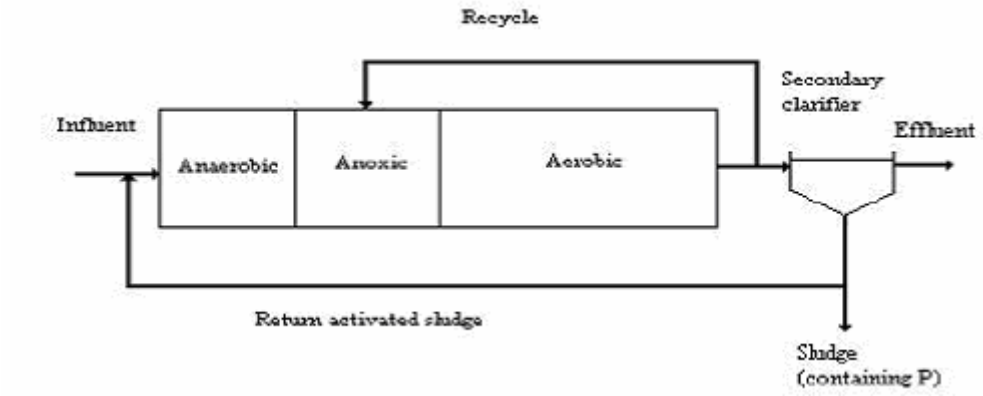


Figure 2.11 Flow diagram of A^2/O process

The Phoredox configuration was combined with the Bardenpho process and resulting new process was termed five-stage modified Bardenpho process capable of simultaneous biological N, P removal. The staging sequence and recycling methods are different from the A^2 / O process. A second anoxic stage is provided for additional denitrification using nitrate produced in the aerobic stage as the electron acceptor, and the endogenous organic carbon as the electron donor. The purpose of final aerobic zone was striping residual nitrogen gas from solution and to minimize the possible release of P in the final sedimentation tanks. The process needs a longer SRT (10 to 20 days) than A^2 / O process, and thus increases the carbon oxidation capability of the organisms.

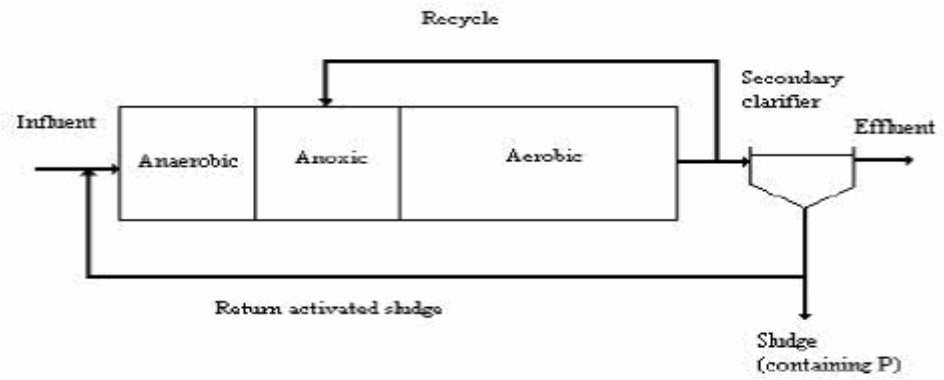


Figure 2.12 Flow diagram of the modified (five-stage) Bardenpho process

Marais and his co-workers determined the adverse effects of electron acceptors entering to the anaerobic zone on the performance of EBPR process. They developed a modification of the three-stage Phoredox process that eliminated the recycle of nitrates in the RAS to the anaerobic zone. This process was named as the University of Cape Town (UCT) process. The UCT process was developed to minimize the electron acceptor input to the anaerobic zones of EBPR processes. The UCT process is similar to the A^2 / O process with two exceptions. The return activated sludge is recycled to the anoxic stage instead of the aerobic stage, and the internal recycle is from the anoxic stage resulting in elimination of nitrate from the anaerobic stage, thereby improving the uptake of phosphorus in the anaerobic stage. The internal recycle feature provides increased organic utilization in the anaerobic stage. The mixed liquor from the anoxic stage contains substantial soluble substrate but negligible nitrate. The recycle of the anoxic mixed liquor provides almost optimal conditions for fermentation uptake in the anaerobic stage. Because the mixed liquor is at a lower concentration, the anaerobic

detention time must be longer than that used in the Phoredox process, and is in the range of 1 to 2 h. The anaerobic recycle rate is typically 2 times the influent flow rate. The UCT process was later modified to provide even better protection of the anaerobic zone from electron acceptors. The return activated sludge was directed to an anoxic reactor that did not receive internal nitrate recycle flow. The nitrate was reduced in this tank, and the mixed liquor from the reactor was recycled to the anaerobic tank. The second anoxic tank follows the first anoxic tank and receives internal nitrate recycle flow from the aeration tank to provide the major fraction of nitrate removal for the process.

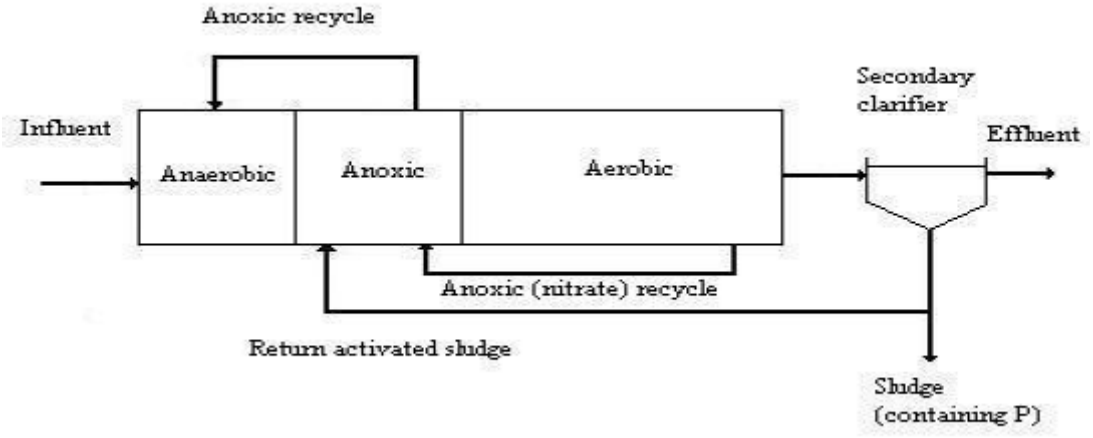


Figure 2.13 Flow diagram of the UCT process

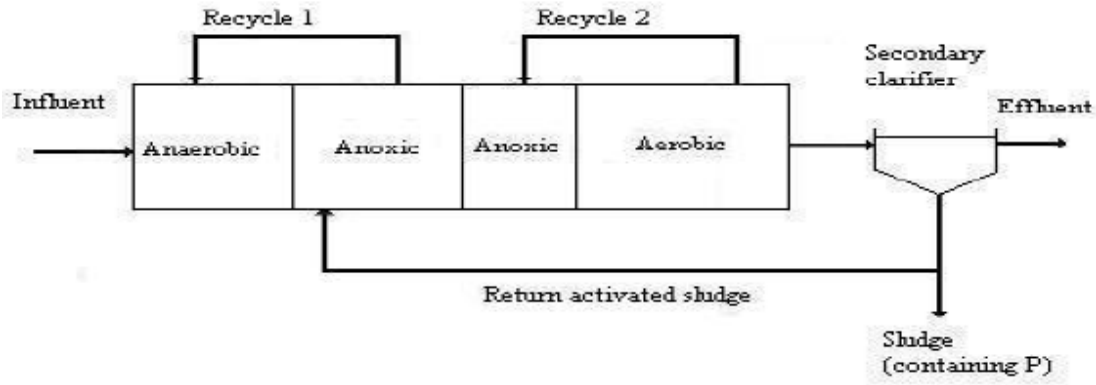


Figure 2.14 Flow diagram of the modified UCT process

Randall (1984) recommended utilization of the UCT configuration as a high-rate BNR system for economical treatment of municipal wastewaters in the U.S.A. This concept was piloted by CH2M/HILL with consultation from Randall for the

Hampton Roads Sanitation District (HRSD), as a new design for replacement of their Lambert' s Point wastewater treatment plant, which was providing only primary treatment (Daigger et al., 1987). The pilot study was very successful, and the process was patented as a public domain patent by CH2M/HILL and HRSD as the Virginia Initiative Project (VIP) process. The VIP process is similar to the A^2 / O and UCT processes except for the methods used for recycle systems. In the VIP process, all the zones are staged consisting of at least two completely mixed cells in series. The return activated sludge is discharged to the inlet of the anoxic zone along with nitrified recycle from the aerobic zone. The mixed liquor from the anoxic zone is returned to the head end of the anaerobic zone. The VIP process is also designed as a high-rate system, operating with much shorter SRTs, which maximize biological phosphorus-removal efficiency. The combined SRT of the anaerobic and anoxic zones is generally 1.5 to 3 d, while the anaerobic and anoxic hydraulic retention time values are typically 60 to 90 min each. The aeration zone is designed for nitrification.

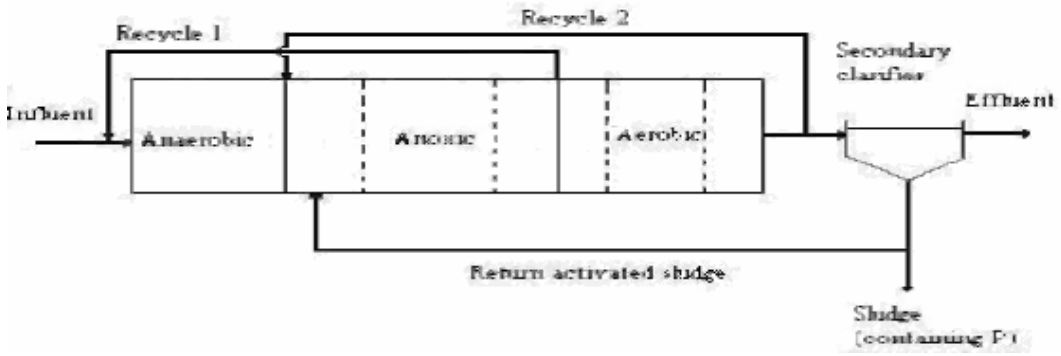


Figure 2.15 Flow diagram of the VIP process.

The process which originated from in Johannesburg, South Africa, is an alternative to the UCT process to minimizing electron acceptor input to the anaerobic zone for the removal of weak wastewater. The return activated sludge is directed to an anoxic zone to reduce the nitrate concentration before feeding to anaerobic zone. The nitrate is reduced by endogenous respiratory reactions and required detention time is dependent on the MLSS concentration, temperature and nitrate concentration of return sludge. In this process configuration, a higher MLSS concentration could be

maintained in the anaerobic zone as compared to the UCT process requiring 1h retention time.

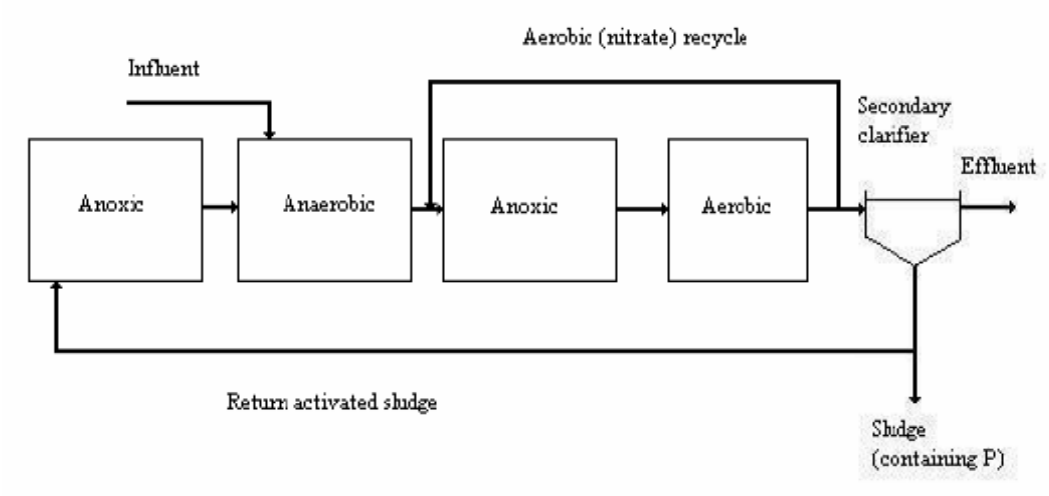


Figure 2.16 Flow diagram of the Johannesburg process.

The sequencing batch reactor (SBR) is a modification of conventional continuous flow activated sludge sewage treatment systems and they could be easily modified for BNR providing anoxic or anaerobic periods within the standard cycles. Because the SBR process operates in a series of timed steps, reaction and settling can occur in the same tank, eliminating the need for a final clarifier. The SBR technology has the advantage of being very flexible in terms of matching react and settle times to the strength and treatment characteristics of a particular waste stream, both nitrogen and phosphorus removal are possible, process is easy to operate, mixed-liquor solids cannot be washed out by hydraulic surges and quiescent settling may produce lower effluent TSS concentration. The nitrate concentration should be minimized before settling and little amount of nitrate is available to compete with rbsCOD in the fill and following reaction period. Thus, anaerobic conditions occur in the fill and initial react period so that rbsCOD uptake and storage by PAOs can occur instead of rbsCOD by denitrifiers.

Table 2.6 Design parameters of EBPR processes (Metcalf & Eddy, 2003)

Design Parameter/Process	**SRT, d**	**MLSS, mg/L**	**Anaerobic Zone (h)**	**Anoxic Zone (h)**	**Aerobic Zone (h)**	**RAS, % of Influent**	**Internal Recycle, % of Influent**
A/O	2-5	3000-4000	0.5-1.5	-	1-3	25-100	-
A^2/O	5-25	3000-4000	0.5-1.5	0.5-1	4-8	25-100	100-400
UCT	10-25	3000-4000	1-2	2-4	4-12	80-100	200-400 (anoxic) 100-300 (aerobic)
VIP	5-10	2000-4000	1-2	1-2	4-6	80-100	100-200 (anoxic) 100-300 (aerobic)
Bardenpho (five-stage)	***10-20***	***3000-4000***	***0.5-1.5***	1-3 ***(1st stage)*** ***2-4*** (2nd stage)	4-12 ***(1st stage)*** ***0.5-1*** (2nd stage)	***50-100***	***200-400***
PhoStrip	5-20	1000-3000	8-12	-	4-10	50-100	10-20
SBR	20-40	3000-4000	1.5-3	1-3	2-4	-	-

CHAPTER THREE
MATERIALS AND METHODS

3.1 Full-scale experimental studies

3.1.1 EBPR configuration of the WWTP

The WWTP, in which these investigations were carried out, was designed to treat an average daily flow rate of 7 m^3/s. Under wet weather conditions, the plant is capable to treat 12 m^3/s. Wastewater first comes to pre-treatment units which consist of fine screens, aerated grit chambers and circular primary sedimentation tanks. Following the grit removal process, wastewater is distributed equally to three treatment lines. On each of these treatment lines, wastewater is settled in primary clarifiers and then introduced to anaerobic tank. In the anaerobic tank, settled wastewater comes into contact with microorganisms carried by the return sludge line as shown in Figure 3.1. Actual design value of HRT_a is 1.1 hours for the average flow rate. During the monitoring period return sludge flow rate was 76% of inflow. HRT_a was calculated using following equation;

$$HRT_a = \left(V_{anaerobic} \Big/ \Sigma Q \right) \tag{3.1}$$

Where, HRT_a is hydraulic retention time in the anaerobic tank, hours

$V_{anaerobic}$ is total volume of anaerobic tank, m^3

ΣQ = Influent flow (m^3/h) + Return sludge flow (m^3/h)

After this anaerobic contact period, the wastewater is exposed to anoxic and aerobic conditions in the oxidation ditches for combined removal of carbon (C), nitrogen (N) and phosphorus (P). Oxidation ditches were designed for having 49% anoxic and 51% aerobic volume. Nitrate is recycled from aerobic zone to anoxic zone with an internal recycle for denitrification. The internal recirculation ratio is 400% of inflow.

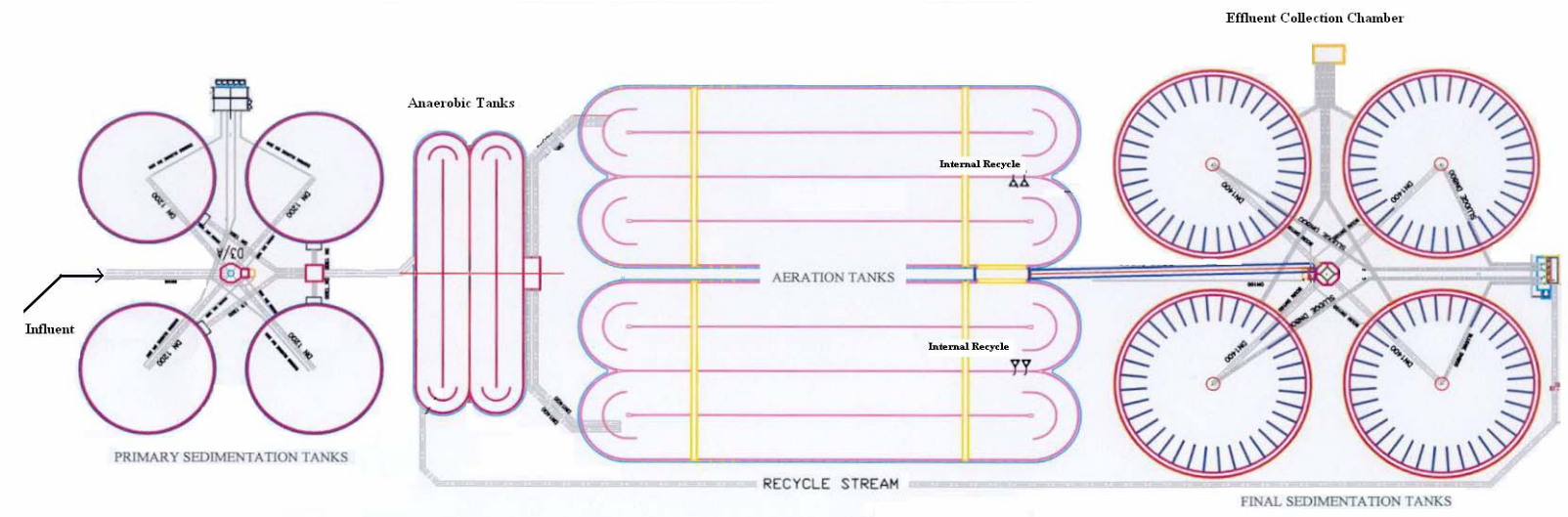


Figure 3.1 EBPR process configuration of the WWTP

Table 3.1 Monitored parameters for establishing mass-balance equations

Influent (Before PS)	Influent of Anaerobic Tank	Effluent of Anaerobic Tank	Anoxic Zone	Aerobic Zone	Effluent	Recycle Stream
COD, mg/L	COD, mg/L	sCOD, mg/L	sCOD, mg/L	sCOD, mg/L	COD, mg/L	sCOD, mg/L
TN, mg/L	sCOD, mg/L	PO_4-P, mg/L	NO_3-N, mg/L	NO_3-N, mg/L	sCOD, mg/L	NO_3-N, mg/L
TP, mg/L	rbsCOD, mg/L	MLVSS, mg/L	PO_4-P, mg/L	PO_4-P, mg/L	rbsCOD, mg/L	PO_4-P, mg/L
PO_4-P	TN, mg/L		MLVSS, mg/L	MLVSS, mg/L	TN, mg/L	MLVSS, mg/L
Flow, m^3/s	TP, mg/L		Internal flow,		NH_4^+-N	External (recycle)
	PO_4-P, mg/L		m^3/s		TP, mg/L	flow, m^3/s
					PO_4-P, mg/L	

DO concentrations is adjusted to 0.2-0.5 in the anoxic and 2 mg/L in the aerobic zone by computer aided online measurement system. Biomass and effluent phases are separated in circular final clarifiers. Settled sludge is collected in a chamber from where it is returned to the anaerobic zone; waste sludge is transferred to the sludge handling facilities.

3.1.2 Sampling methods

Flow proportional 2-h composite samples were collected periodically. Average flow rate during the 2-h sampling period was used in calculations. Samples were diluted with distilled water according to the measurement range of cell test. As chemical characteristics of the samples (both wastewater and activated sludge) are very unstable, the period from sampling to analysis were kept as short as possible. Especially due to large WWTP sampling area, to obtain more accurate analysis results, soluble nutrient forms such as PO_4-P, sCOD, NO_3-N etc. were filtered immediately at the sampling points using both rough filter paper and single use syringe filter that has a pore size of 0,45 μm. (Sartorius® Minisart RC 25). Although chemical parameters including suspended materials such as COD, Total Phosphorus (TP), and Total Nitrogen (TN) are less sensitive to rapid changes, they were also analyzed immediately after sampling without filtration. These samples were digested using microwave digestion unit (Merck® MW 520) and its original digestion reagents. P content of activated sludge on MLVSS basis was determined by microwave digestion method after 1:10 dilution with distilled water according to Merck user's manual.

3.1.3 Analytical methods

BOD_5 measurements were performed by respirometric methods in which pressure difference were measured automatically by the aid of lithium hydroxide. Wastewater samples were prepared according to the DIN 38409-52 standard and nutrient level of the wastewater was adjusted by commercial BOD nutrient buffer pillows. Nitrification reactions were inhibited using N - cllythiarea solution (5 g L^{-1}).

Temperature was kept constant during the 5-days incubation period at 20±0.1 ˚C by automatic control system.

COD concentration between 150 – 1000 mg L^{-1} was determined according to the DIN 38409 – H41 – H44 standard. The method is based on reaction of oxidizable substances with sulphiric acid – potassium dichromate solution in the presence of silver sulphate as a catalyst. In this method chloride interference is eliminated by mercury sulphate. The green coloration of Cr^{3+} is then determined photometrically.

COD concentration between 15 – 150 mg L^{-1} was determined according to the DIN 38409 – H41 – H44 standard. The principle of the method could be explained with reaction between oxidizable substances with sulphiric acid – potassium dichromate solution in the presence of silver sulphate as a catalyst. Chloride is masked by mercury sulphate. The reduction in the yellow coloration of Cr^{6+} is then determined photometrically.

VFAs concentration was determined according to the DIN 38409-H16 standard. This method is based on reaction between fatty acids and diols in an acidic environment, resulting with formation of fatty acid esters. These compounds are reduced by iron (III) salts to form red colored complexes which could be measured photometrically.

Total nitrogen concentration was determined according to the EN ISO 11905-1 standard by photometric measurement. The principle of the measurement method could be summarized that inorganically and organically bonded nitrogen is oxidized to nitrate by digestion with peroxide-sulphate. The nitrate ions then reacts with 2.6-dimethylphenol in a solution containing sulphiric and phosphoric acid to form nitrophenol.

Ammonium concentration was determined according to the DIN 38406-E5 standard. The method is based on reaction of ammonium ions with hypochlorite and

salicylate ions in the presence of sodium nitroprusside as a catalyst to form indophenols blue at pH 12.6.

Nitrate concentration was determined according to the EN ISO 38405-D9-2 standard. The principle of the method could be explained that nitrate ions in solution react with 2.6- dimethylphenol to form 4-nitro 2.6 – dimethylphenol by the aid of sulphiric and phosphoric acids.

Total phosphorus and phosphate concentration were determined according to the EN ISO 1189 standard by photometric measurement. The principle of the method is based on reaction of phosphate ions with molibdat and antimony in an acidic solution to form antimony phosphomolibdat complex, which is reduced by ascorbic acid to phospho-molybdenum blue which could be measured photometrically.

Suspended solids (SS), mixed liquor suspended solids (MLSS) and MLVSS were measured according to Standard Methods (APHA-AWWA-WPCF, 1998). pH and temperature were measured using well calibrated manual probes from WTW®.

3.1.4 Determination of biomass (MLVSS) P content

To measure P content of MLVSS, some part of the freshly collected activated sludge sample was filtrated on site immediately after sampling using 0.45 μm syringe filter. Reaming part of the activated sludge sample was diluted at 1:10 ratio with distilled water. 5 ml of diluted activated sample were digested after addition of required original digestion reagents. For the digestion process, Merck® MW 520 digestion unit was used. Phosphate concentration of the on site - filtered sample and digested sample were measured. For the analysis of digested activated sludge and on site – filtered sample, 1.14848.0001 phosphate measurement cell test were used. Unfiltered activated sludge was kept under aerobic conditions for the time period between sampling and analysis. P content of MLVSS as percentage was calculated by the following equation;

$$\text{MLVSS P Content, \%} = \left(\frac{[PO_4 - P_{ad}] - [PO_4 - P_{af}]}{[\text{MLVSS}]} \right) \times 100 \qquad (3.2)$$

Where, $[PO4\text{-}P_{ad}]$ is PO4-P concentration after digestion, mg/L

$[PO4\text{-}P_{af}]$ is PO4-P concentration after filtration, mg/L

[MLVSS], mixed liquor suspended solids concentration in aerobic zone, mg/L

3.1.5 Establishing mass - balance equations

3.1.5.1 Determination of nutrient levels influent of the anaerobic tank

The return of activated sludge from the final sedimentation tanks to the inlet of the anaerobic tank is essential feature for the maintenance of desired level of microorganisms in the process. A simplified diagram used in establishing mass - balance is given in the Figure 3.2.

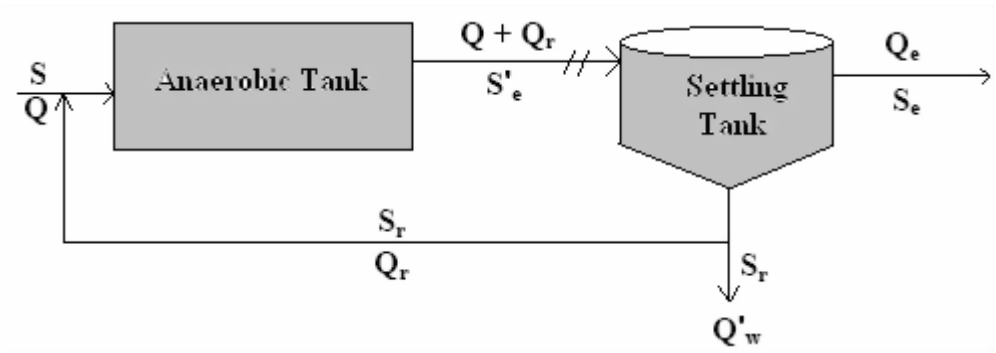


Figure 3.2 Simplified diagram used in establishing materials balance

As it could be seen from Figure 3.2, the concentrations of the nutrients in the inlet of the anaerobic tank should be determined using materials balance considering both influent characteristics and the return activated sludge characteristics (as both flow and concentration) (Metcal & Eddy, 1991). The resulting expression could be defined as follows;:

$$Q(S)+Q_r(S_r)=(Q+Q_r)\times S_f \quad (3.3)$$

Solving the equation 3.3 for S_f yields:

$$Sf = [Q(S)+Q_r(S_r)] / [(Q+Q_r)] \quad (3.4)$$

Where, S*f* is balanced influent nutrient concentration, mg/L
Including: COD, sCOD, rbsCOD, TN, NO_3-N, TP, PO_4-P, DO
Q is influent flow rate, m^3/s
Q_r is return sludge flow rate, m^3/s
S is influent nutrient concentration
S_r is return sludge nutrient concentration

3.1.5.2 Determination of PO_4-P concentration in the influent of anoxic zone

Figure 3.3 illustrates the denitrification process used in EBPR. In this process configuration, nitrified mixed liquor is recycled to denitrification zone. The internal recycle ratio is generally in the range of 4:1 based on the influent flow.

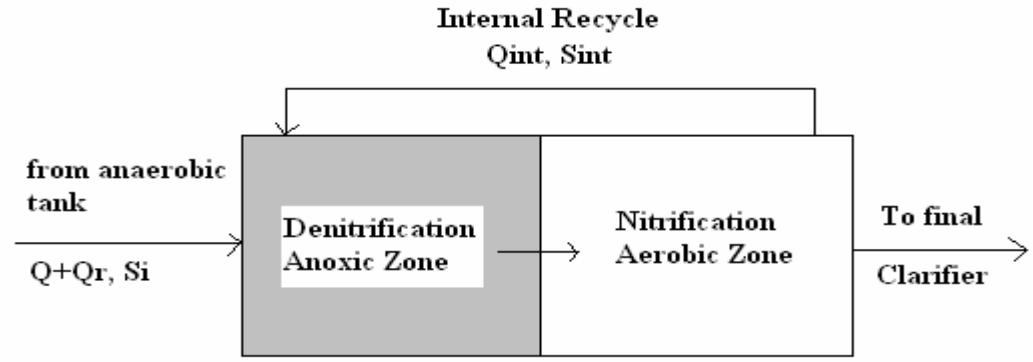


Figure 3.3 Simplified diagram used in establishing materials balance around anoxic zone

In this zone, substrates forms in the influent are used as electron donor and nitrate is used as electron acceptor (USEPA, 1993). Soluble substrate forms in the anoxic zone could be determined performing mass-balance, which results in the following equation rearranged for PO_4-P:

$$\left[\Sigma PO_4 - P_{anx-\inf}\right] = \frac{\left[(Q+Q_r)\times\left[PO_4 - P_{an-eff}\right] + \left(Q_{\text{int}}\times\left[PO_4 - P_{ae}\right]\right)\right]}{\left[(Q+Q_r+Q_{\text{int}})\right]} \quad (3.5)$$

Where, $[\Sigma PO_4\text{-}P_{anx\text{-}inf}]$ is balanced PO_4-P concentration in anoxic zone, mg/L
Q is influent flow rate, m^3/s
Q_{int} is internal recycle flow rate, m^3/s
Q_r is return sludge flow rate, m^3/s
$[PO_4\text{-}P_{an\text{-}eff}]$ is PO_4-P concentration in the anaerobic tank effluent, mg/L
$[PO_4\text{-}P_{ae}]$ is PO_4-P concentration in the aerobic zone, mg/L

3.1.5.3 Determination of nitrate concentration in aerobic zone

Denitrification is one of the important considerations in EBPR. It could be assumed that the influent total nitrogen converted to nitrate, effluent ammonia and solids synthesis. The amount of nitrogen used in synthesis could be determined from COD removed. The nitrogen content of bacteria is 10-12 percent but the presence of inert and nonbiodegradable solids a value of 5-8 percent based on dry substances could be more accurate. Establishing a mass balance for a configuration given in Figure 3.3, amount of TN converted to nitrate, solids synthesis and effluent ammonia could be expressed in following expression; (USEPA, 1987)

$$NO = N_o - NH_e - N_{syn} \quad (3.6)$$

Where, NO is total nitrogen to be denitrified, mg/L
No is influent total nitrogen, mg/L
NH_e is effluent ammonium concentration, mg/L
N_{syn} is nitrogen used in solids synthesis, mg/L

$$N_{syn} = Y_n \times COD_{removed} \times F_n \quad (3.7)$$

Where, Y_n is yield coefficient, gVSS/gCOD
$COD_{removed}$ is removed COD in the biological treatment , mg/L
F_n is fraction of nitrogen in MLVSS, %

After determining the NO, performing a mass balance nitrate concentration in the aerobic zone (N) could be estimated as follows;

$$N = NO/(R + r + 1) \tag{3.8}$$

Where, NO is as defined before, mg/L
R is internal recycle ratio
r is return sludge ratio

3.1.5.4 Determination of PAOs phosphorus content in anoxic zone

The nitrate entering to the anoxic zone is utilized through the denitrification reactions. Total nitrate removed in the anoxic zone could be calculated as follows;

$$[NO_3 - N_{removed}] = [NO] - [NO_3 - N_{eff}] \tag{3.9}$$

Where, $[NO_3\text{-}N_{removed}]$ is removed nitrate concentration in the anoxic zone, mg/L
[NO] is as defined before, mg/L
$[NO_3\text{-}N_{eff}]$ is effluent nitrate concentration, mg/L

At the second step, biomass production as a result of nitrate removal could be determined by following equation (Metcalf & Eddy, 2003):

$$Px_{PAOs-anoxic} = Y_{dn} \times [NO_3 - N_{removed}] \tag{3.10}$$

Where, $Px_{PAOs\text{-}anoxic}$ is biomass production as a result of nitrate removal, $gVSS/m^3$
Y_{dn}, Denitrifier synthesis yield, gVSS/d NO_3-N

$[NO_3\text{-}N_{removed}]$ is as defined before, mg/L

$$[PO_4 - P_{rem\text{-}anoxic}] = \Sigma[PO_4 - P_{anoxic\text{-}inf}] - [PO_4 - P_{eff}] \quad (3.11)$$

Where, $[PO_4\text{-}P_{rem\text{-}anoxic}]$ is removed concentration of PO_4-P in anoxic zone, mg/L

$\Sigma[PO_4\text{-}P_{anoxic\text{-}inf}]$ is balanced initial concentration of PO_4-P in the anoxic zone, mg/L

$[PO_4\text{-}P_{eff}]$ is effluent PO_4-P concentration, mg/L

It was reported by Metcal & Eddy, (2003) that concentration of removed PO_4-P could also be defined as follows;

$$[PO_4 - P_{rem\text{-}anoxic}] = Px_{PAOs\text{-}anoxic} \times F_p \quad (3.12)$$

Where, $[PO_4\text{-}P_{rem\text{-}anoxic}]$ is as defined before, mg/L

$Px_{PAOs\text{-}anoxic}$ is as defined before, gVSS/m^3

F_p, Fraction of P in PAOs

Fraction of P in PAOs could be determined as percentage by solving equation (3.12) for F_p:

$$F_p = \left[[PO_4 - P_{rem\text{-}anoxic}] / Px_{anoxic} \right] \times 100 \quad (3.12)$$

3.1.5.5 Determination of PAOs and the activated sludge phosphorus content in aerobic zone

According to the theory of EBPR, rbsCOD utilization resulting with acetate uptake is very important in determining the produced amount of PAOs. If significant amount of electron acceptors input to the anaerobic zone, the acetate could be depleted before it is taken up by PAOs. Since rbsCOD could be converted to acetate easily in shot anaerobic retention times, influent rbsCOD level is very important in determining the quantity of P removed by the biological storage. Investigations showed that low influent rbsCOD concentrations rapidly lead to deteriorated EBPR efficiency. The amount of biologically removed P could be estimated from the

amount of rbsCOD available in influent. According to the stoichiometry of EBPR, cell P content is 0.3 g P/VSS and 10 g of rbsCOD is required to remove 1 mg of P through the biochemical pathways of EBPR. Removal of bpCOD is results with additional P removal by normal cell (non-PAOs) synthesis (Metcal&Eddy, 2003).

Using the above given stoichiometry, total cell synthesis in the EBPR process could be defined as follows:

$$\Sigma Px = \Sigma Px_{PAOs} + \Sigma Px_{non-PAOs} \tag{3.13}$$

Where, ΣPx is total biomass production, gVSS/m^3
ΣPx_{PAOs} is biomass production from rbsCOD utilization (PAOs cell synthesis), g/m^3
$\Sigma Px_{non\text{-}PAOs}$ is biomass production from bpCOD utilization (non - PAOs cell synthesis), gVSS/m^3

PAOs cell synthesis could be divided into two groups according to the used electron acceptors which composed of nitrate ($PAOs_{anoxic}$) and DO ($PAOs_{aerobic}$) (Brdjanovic et. al, 1997; Lee et. al., 2003).

$$\Sigma Px_{PAOs} = Px_{PAOs-anoxic} + Px_{PAOs-aerobic} \tag{3.14}$$

Solving equation (3.14) in which the variable as defined before for $Px_{PAOs\text{-}aerobic}$ yields:

$$Px_{PAOs-aerobic} = \Sigma Px_{PAOs} - Px_{PAOs-anoxic} \tag{3.15}$$

It could be assumed that total PAOs cell synthesis happens due to rbsCOD consumption in the anaerobic zone. To obtain accurate results, the amount of rbsCOD used in denitrification reactions by the aid of electron acceptors (from both return sludge and influent) in aerobic zone is subtracted from the total amount of sCOD depleted in the anaerobic period to estimate the net rbsCOD used by PAOs. Using this theoretical background, total PAOs cell synthesis could be defined as follows (Metcalf & Eddy, 2003);

$$\Sigma Px_{PAOs} = \left[\frac{Y}{1+(k_d)SRT}\right] \times rbsCOD \qquad (3.16)$$

Where, ΣPx_{PAOs} is as defined before, g/m^3
Y is heterotrophic synthesis yield, (assumed as 0.40) gVSS/gCOD
k_d is endogenous decay coefficient, (assumed as 0.08) gVSS/gVSS.d
SRT_{ae} is aerobic solids retention time, days
rbsCOD is soluble readily biodegradable COD, mg/L

Solids retention time could be estimated using the below given equation (Metcalf & Eddy, 2003). :

$$SRT = \frac{VX}{(Q-Q_w)X_e + Q_w X_R} \qquad (3.17)$$

Where, V is reactor volume, m^3
X is MLVSS concentration in aeration tank, g/m^3
Q is inflow, m^3/d
Q_w is waste sludge flow rate, m^3/d
X_e is the concentration of biomass in effluent, g/m^3
X_R is MLVSS concentration of return sludge, g/m^3

Ordinary heterotrophic bacteria (non-PAOs) synthesis from the utilization of pbCOD could be calculated using following equation in which variables as defined before expect for pbCOD (particulate biodegradable COD):

$$\Sigma Px_{PAOs} = \left[\frac{Y}{1+(k_d)SRT}\right] \times pbCOD \qquad (3.18)$$

Particulate biodegradable COD could be estimated following steps (Metacalf & Eddy, 2003):

$$pbCOD = \Sigma bCOD - \Sigma rbsCOD \tag{3.19}$$

Where, pbCOD is particulate biodegradable COD, mg/L
ΣbCOD is balanced concentration of influent total biodegradable COD, mg/L
ΣrbsCOD is balanced concentration of influent total soluble readily biodegradable COD, mg/L

It was reported that BOD data are required to determine the total biodegradable COD (bCOD). The bCOD/BOD ratio is greater than the ultimate BOD (UBOD) to BOD ratio (UBOD/BOD) because not all of the bCOD is oxidized in the BOD test. The bCOD could be determined by following equation:

$$\frac{bCOD}{BOD} = \frac{UBOD/BOD}{1.0 - 1.42 \times f_d \times (Y_h)} \tag{3.20}$$

In this equation the fraction of cell mass remaining as cell debris, g/g (f_d) is assumed as 0.15, synthesis yield coefficient for heterotrophic bacteria (Y_h), gVSS/gCOD is assumed 0.4 and UBOD/BOD ratio is assumed 1.5 to estimate the bCOD.

To determine the P content of aerobic PAOs, the concentration of P luxury uptake was determined using following equation:

$$\Sigma PO_4 - P_{luxuryuptake} = \Sigma PO_4 - P_{anoxic} - \left(PO_4 - P_{eff} + PO_4 - P_{rem-part}\right) \tag{3.21}$$

Where, ΣPO_4-P_{anoxic} is measured PO_4-P concentration in anoxic zone, mg/L
PO_4-P_{eff} is PO_4-P concentration in the effluent, mg/L
PO_4-$P_{rem\text{-}part}$ is concentration of PO_4-P removed during particulate COD utilization, mg/L

PO_4-P removed during pbCOD removal (PO_4-$P_{rem\text{-}part}$) could be estimated using following equation in which heterotrophic bacteria cell P content is assumed as 2% based on MLVSS (or F_p=0,02 g/gVSS) (Metcalf & Eddy, 2003). This assumption

(0.2 g P/gVSS) was also proved by the measurement results conducted on the activated sludge samples taken from the effluent of anaerobic zone since all P is released back to the liquid interface.

$$PO_4 - P_{rem-part} = Px_{nonPAOs} \times F_p \tag{3.22}$$

After determining all of the required variables, P content of aerobic PAOs could be estimated by the following equation as percentage:

$$Fp_{PAOs-aerobic} = \left[\frac{\Sigma PO_4 - P_{luxuryuptake}}{Px_{PAOs-aerobic}} \right] \times 100 \tag{3.23}$$

Since the activated sludge could be classified as non-PAOs and PAOs, P content of the activated sludge is the average of P content of these microorganisms. Therefore P content of the activated sludge could be defined using the following equation in which parameters were as defined before:

$$P_{activatedsludge} = \frac{Px_{p_{bCOD}} \times Fp_{nonPAOs} + Px_{PAOs} \times Fp_{PAOs}}{Px_{nonPAOs} + Px_{PAOs}} \tag{3.24}$$

Intracellular P fraction of the activated sludge was determined using both direct analytical measurement technique and stoichiometrical approach to validate the obtained results. Therefore this approach also provided with the control of estimated microbial mass fractions and intracellular P content.

Another bacterial group that is able to assimilate simple substrate forms in the anaerobic zone of the EBPR process is glycogen accumulating microorganisms (GAOs) (Whang & Park, 2001; Lui et. al., 1997). In this approach, presence of these microorganisms was not directly considered in the stoichiometric equations. However PAOs specific yield coefficient was assumed as 0.40 mg PAOs mg $(rbsCOD)^{-1}$ in the stoichiometric equations (Metcalf & Eddy, 2003).

3.1.6 Determination of available anaerobic reaction time (AART)

Presence of electron acceptors in the anaerobic zone has a net reducing effect on both carbon source and AART (available anaerobic reaction time) via denitrification reactions. In order to investigate the effect of electron acceptors on EBPR efficiency in the anaerobic zone, DO was converted to nitrate as concentration using a theoretical value of $2.86gO_2/gNO_3$-N (USEPA, 1987; Metcalf and Eddy, 2003). HRT_a was then corrected considering denitrification reactions in the anaerobic zone using a nitrate reduction rate. AART was defined as corrected HRT_a according to anaerobic denitrification time. AART was calculated using following equation.

$$AART = HRT_a - T_d \tag{3.25}$$

Where, AART is available anaerobic retention time period after denitrification reactions , hours
HRT_a is as defined before
T_d is required denitrification time in anaerobic period, hours

T_d was calculated using following equation;

$$T_d = \left(\Sigma[NO_3 - N] \Big/ (q_{dn} \times MLVSS_{anaerobic}) \right) \tag{3.26}$$

Where, $\Sigma[NO_3\text{-}N]$ is balanced (superposed) concentration of nitrate in the entrance of anaerobic tank, mg/L
q_{dn} is denitrification rate, $mgNO_3$-N/gVSS/h
$[MLVSS]_{anaerobic}$ is mixed liquor volatile suspended solids concentration in anaerobic tanks, mg/L

3.1.7 Determination of anaerobic sCOD utilization rate

Some part of the sCOD in the wastewater is utilized by denitrifiers by the aid of electron acceptors in the anaerobic zone of EBPR process. Therefore, total amount of sCOD used in P release should be calculated by subtracting sCOD consumed in

anaerobic denitrification reactions from total amount of sCOD used during the anaerobic period. To determine the quantity of sCOD utilized in anaerobic denitrification, 5 mg sCOD/NO_3-N ratio was used (USEPA, 1987).

$$[sCOD]_{net} = [sCOD]_t - [sCOD]_d \quad (3.27)$$

Where, $[sCOD]_{net}$ is sCOD used in P release, mg/L
$[sCOD]_t$ is total amount of sCOD used during anaerobic period, mg/L
$[sCOD]_d$ is sCOD used in denitrification reactions, mg/L

$$q_{sCOD} = \left([sCOD]_{net} \right) / \left([MLVSS]_{anaerobic} \times AART \right) \quad (3.28)$$

Where, q_{sCOD} is anaerobic sCOD utilization rate, mgsCOD/gVSS/h
$[sCOD]_{net}$ as defined before, mg/L
AART is as defined before , hours
$[MLVSS]_{anaerobic}$ is as defined before, g/m^3

3.1.8 Determination of P Uptake and Release Rates

Since several streams having different flow rate and phosphate concentrations combined in the same tank, net amount of $P_{release}$ and P_{uptake} were determined using mass balance equations. In the anaerobic tank, influent wastewater and return sludge is combined. Therefore phosphate concentration and flow rates of influent and recycle sludge were used in calculation of net P release.

$$[PO_4 - P]_r = [PO_4 - P]_e - \Sigma[PO_4 - P]_i \quad (3.29)$$

Where, $[PO_4\text{-}P]_r$ is net amount of P release, mg/L
$[PO_4\text{-}P]_e$ is measured amount of P in the effluent of anaerobic tank, mg/L
$\Sigma[PO_4\text{-}P]_i$ is balanced concentration of P in the influent of anaerobic tank, mg/L

$P_{release}$ rate was calculated using following equation;

$$q_{PO4\text{-}Prel} = \left(\frac{[PO_4 - P]_r}{[MLVSS]_{anaerobic} \times AART} \right) \quad (3.30)$$

Where, $q_{PO4\text{-}Prel}$ is P release rate, mgP/gVSS/h
$[PO_4\text{-}P]_r$ is as defined before, mg/L
$[MLVSS]_{anaerobic}$ is as defined before, g/m^3
AART is as defined before , hours

P_{uptake} rate was calculated using following equations;

$$[PO_4 - P]_u = [PO_4 - P]_e - [PO_4 - P]_i \quad (3.31)$$

Where, $[PO_4\text{-}P]_u$ is net amount of P uptake, mg/L
$[PO_4\text{-}P]_e$ is measured P concentration in the effluent of anaerobic period, mg/L
$[PO_4\text{-}P]_i$ is measured amount of P in the effluent (plant effluent), mg/L

$$q_{PO4\text{-}Pup} = \left([PO_4 - P]_u \Big/ ([MLVSS]_{aerobic} \times HRT_{ae}) \right) \quad (3.32)$$

Where, $q_{PO4\text{-}Pup}$ is P uptake rate, mgP/gVSS/h
$[PO_4\text{-}P]_u$ is as defined before, mg/L
$[MLVSS]_{aerobic}$ is mixed liquor volatile suspended solids concentration in aerobic zone, g/m^3
HRT_{ae} is hydraulic retention time in the aeration tank, hours

$P_{uptake}/P_{release}$ Ratio was calculated using following equation;

$$P_{uptake}/P_{release}\ \text{Ratio} = \left(\frac{[PO_4 - P]_u}{[PO_4 - P]_r} \right) \quad (3.33)$$

Where, $[PO_4\text{-}P]_u$ is as defined before, mg/L
$[PO_4\text{-}P]_r$ is as defined before, mg/L

3.2 Experimental set-up for batch tests

2 L Woulf Bottles (Woulff'sche-Flaschen, DURAN®, Schott) were used to perform the batch-scale experiments. During the batch tests, mixing was provided by a magnetic stirrer. N_2 gas was fed to the reactor by a diffuser from N_2 tube in anaerobic and anoxic batch tests. Oxygen demand was provided with an air pump connected to a diffuser in aerobic batch tests. All samples were filtered immediately using single use syringe filter that has a pore size of 0.45 μm. Activated sludge characteristics of different sludge samples were examined by batch tests. Different EBPR process configurations used in cultivation of activated sludge samples were demonstrated in Figure 3.4 and Figure 3.5. Similar test procedures were also used to compare seasonal activated sludge characterizations. Procedures of applied batch tests were explained in following sections. Together with batch tests, the P removal performance and characteristics of two different treatment lines were also examined full-scale tests using above given procedures.

3.2.1 Anaerobic phosphorus release batch test

Freshly collected 2 L of effluent and 2L of activated sludge samples were obtained from the treatment plant return sludge pumping station. A portion of the effluent, acetate and activated sludge were placed into the reactor. Volumetric mixture ratio between effluent, acetate and activated sludge were determined according to the F/M ratio of the treatment plant. The reactor was continuously flushed by N_2 gas to create anaerobic conditions. Samples were taken periodically at 1, 5, 10, 15, 20, 30, 60 and 90 minutes. Experimental set up used in anaerobic P release batch tests was shown in Figure 3.4.

Acetate in excess (100-500 mg HAc-C /L or (266-800 mg sCOD/L) was instantly added to the reactor under anaerobic conditions. PO_4-P, VFAs and MLVSS concentrations were measured for 1.5 hours to determine the phosphorus release rate (mg P / g VSS min) and VFAs consumption rate (mg VFAs / g VSS min). At the end of the test, surplus VFAs always remained in solution to be sure that carbon was not limited phosphorus release (Bradjanovic, 1998).

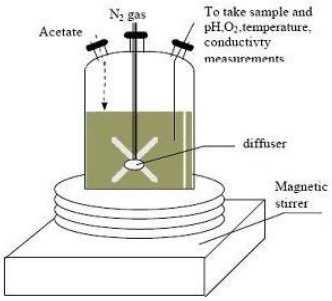


Figure 3.5 Experimental set up for anaerobic P release batch test

3.2.2 Aerobic phosphorus uptake batch test

The sludge exposed first to anaerobic conditions in presence of acetate in order to deplete the internal poly-P pool and increase the PHA level of the biomass so that PHB does not limit the P-uptake process. Acetate (20-25 mg HAc-C/L) was instantly added to the reactor under anaerobic conditions. After 3 h, acetate was fully consumed and phosphorus was released into solution. At this point half of the sludge was transferred to another, identical, reactor. The remaining part of the sludge was exposed to aerobic conditions to measure the maximal specific aerobic phosphorus uptake rate. In case the surplus acetate remains in solution after the anaerobic phase, the sludge should be washed and phosphate should be manually added to the mixed liquor prior to splitting (Brdjanovic, 1998). Lab-scale batch reactor was assembled as shown in Figure 3.6.

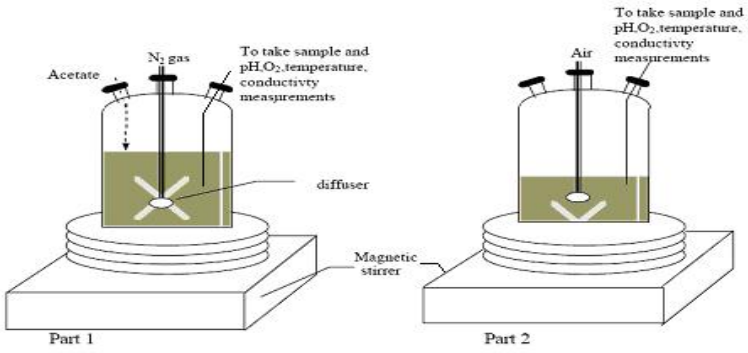


Figure 3.6 Experimental set up for aerobic P uptake batch test

The reactor was mixed for 3.5 hours and samples were taken periodically at 0, 1, 30, 60, 90, 120, 180 and 210 minutes. PO_4-P and MLVSS concentrations were measured to determine the aerobic phosphorus uptake rate as mg P / g VSS min.

3.2.3 Anoxic phosphorus uptake batch test

The second part of the sludge cultivated in the foregoing batch test was exposed to anoxic conditions in order to determine the maximal specific anoxic phosphorus uptake rate (Brdjanovic, 1997). Anoxic conditions were maintained by addition of surplus amount of nitrate at the beginning of the test (28 mgN/L) and by the continuous flushing of the mixed liquor by N_2 gas. Lab-scale batch reactor was assembled as shown in Figure 3.7.

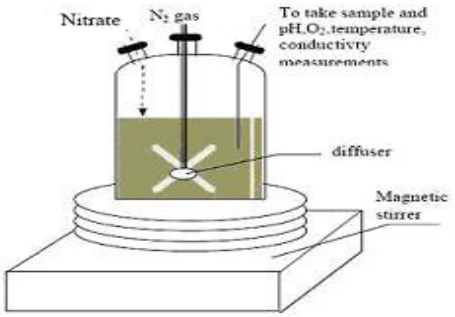


Figure 3.7 Experimental procedure for anoxic P uptake batch test

This batch test, which was similar to the procedure used for Aerobic Phosphorus Uptake Batch Test. Excess amount of nitrate (28 mg/N) was supplied from 10 g/L stock KNO_3 solution to create anoxic conditions. N2 gas was introduced into the reactor with a diffuser. Mixing was provided by a mechanical stirring. The reactor was mixed for 210 minute and samples were taken periodically at 0, 1, 30, 60, 90, 120, 210 minutes. NO_3-N and MLVSS concentrations were measured to determine the anoxic phosphorus uptake rate (mg P /g VSS min) and nitrate utilization rate (mg NO_3-N g VSS min)

3.3 Statistical analysis

Some part of the experimental results (MLVSS P content, sludge age, flow rates etc.) obtained from full-scale studies, discussed in the Chapter 6, 7, were statistically examined using Independent Samples T Test at 0.95 confidence interval. SPSS v.13 software was used in the data analysis. Evaluation of the statistical analysis results was performed using the user's manual of the SPSS v13 software. It was explained

that The Independent-Samples T Test could be used to compare the means of two groups. If the significance value for the Levene test is high (typically greater than 0.05) the results that was obtained assuming equal variances for both groups could be used. If the significance value for the Levene test is low the results that do not assume equal variances for both groups should be used. A low significance value for the t test (2 tailed - typically less than 0.05) could be safely concluded that there is a significant difference between the two group means. If the confidence interval for the mean difference does not contain zero, this also an indication of is a significant difference between the two group means. If the significance value is high and the confidence interval for the mean difference contains zero, then it could not be concluded that there is a significant difference between the two group means (SPSS v13 Manuel, 2004).

Determination of the relationship between data groups, their regression types and determination coefficients were performed using M.S. Excel, 2003 software.

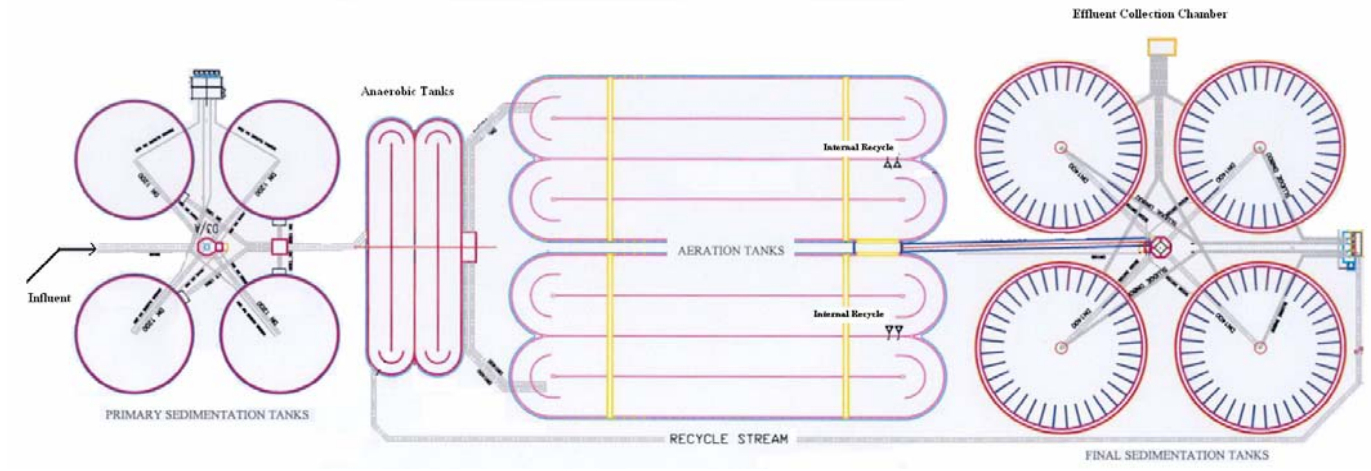


Figure 3.4 Operation of the EBPR process configuration with primary sedimentation

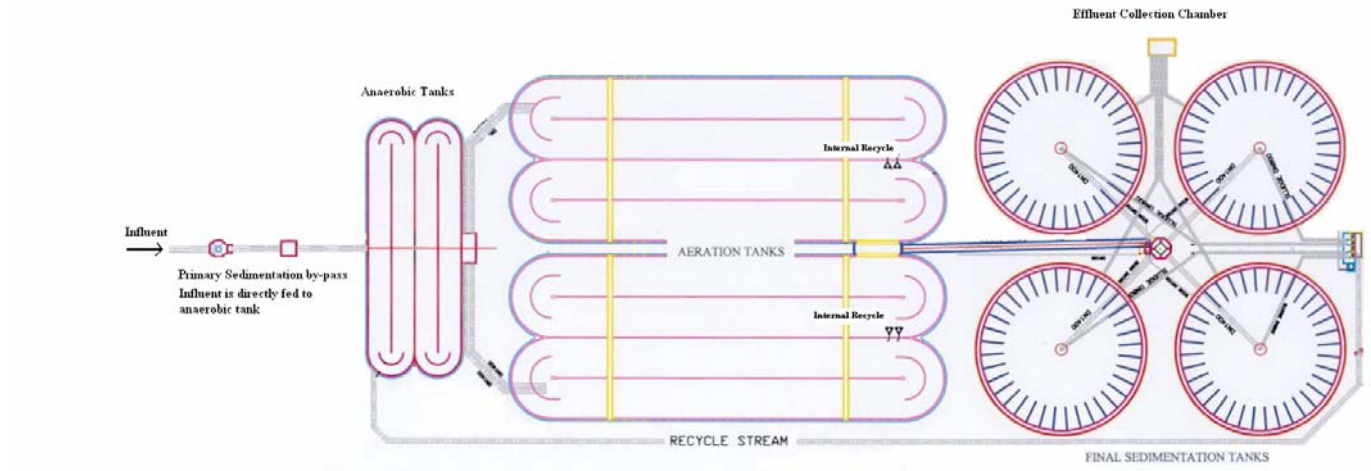


Figure 3.5 Operation of the EBPR process configuration without primary sedimentation

CHAPTER FOUR
SIMULTANAEUS EFFECT of ANAEROBIC RETENTION TIME & PRESENCE of ELECTRON ACCEPTORS in ANAEROBIC ZONE ON BIOLOGICAL PHOSPHORUS REMOVAL CHARACTERISTICS of LARGE SCALE WASTEWATER TREATMENT PLANT

4.1 Wastewater characterization

A reliable wastewater characterization is essential for design, prediction of effluent quality and optimization of the existing plants. Flow characterization is also very important in understanding the behavior of a plant in combination with the chemical and physical properties of the wastewater. Carbonaceous constituents, especially rbsCOD concentration of wastewater have significant effect on EBPR. Since rbsCOD can easily be converted to acetate via fermentation and assimilated into PAOs in the anaerobic zone, it is the most critical parameter in accurate evaluation of EBPR processes (Metcalf and Eddy, 2003; Park et al., 1997).

Flow measurement results showed that there was a significant variation, mainly affecting the HRT_a. Typical daily, dry weather flow characteristic of the WWTP is shown in Figure 4.1. This figure presents monthly average flow data and HRT_a occurring at average flow. It was reported that, the optimum HRT_a varies according to the process type and should be higher than 1 hour for many EBPR processes (USEPA 1987; Metcalf & Eddy 2003). It is evident from Figure 4.1 that HRT_a was lower than 1 hour lasting for 7 hours.

Typical composition of influent wastewater recorded during the monitoring period are summarized in Table 4.1 Environmental parameters including pH and temperature were good enough to obtain a stable EBPR. One of the critical parameter measured during the monitoring period was DO concentration. It could be seen from Table 1 that there were significant DO input to the anaerobic tank.

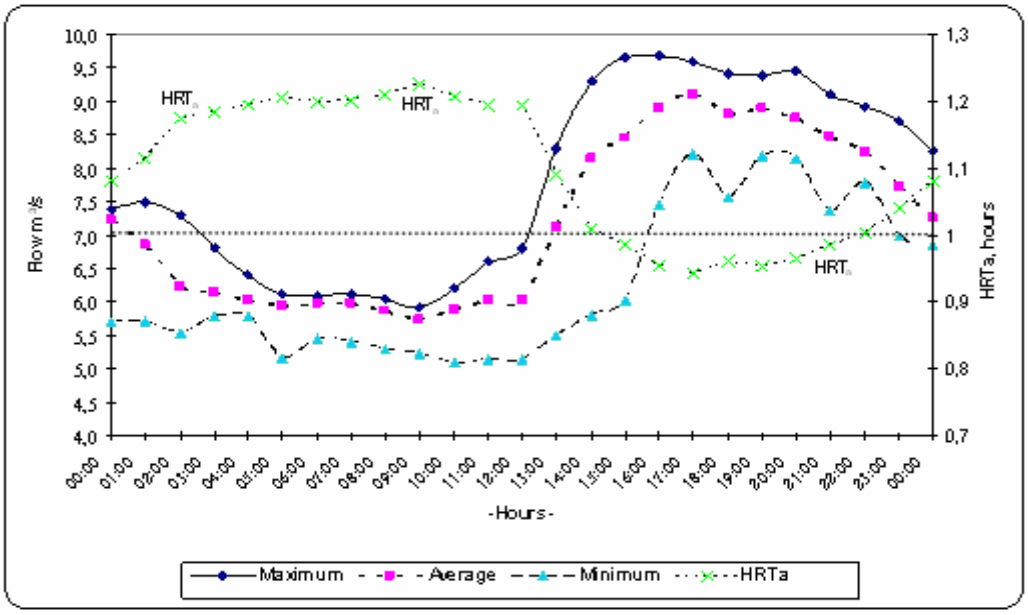


Figure 4.1 Typical daily flow variation of the WWTP (Dry weather conditions)

4.2 Operational conditions maintained in the monitoring period

Average sludge age and temperature were 14 days and 18 °C. It could be said that sludge age was also in acceptable limits for EBPR. Measured MLVSS concentrations in anaerobic, aerobic zone and in return sludge were depicted in Fig. 4.2. This figure also indicates that all environmental and operational parameters were suitable for microbial growth.

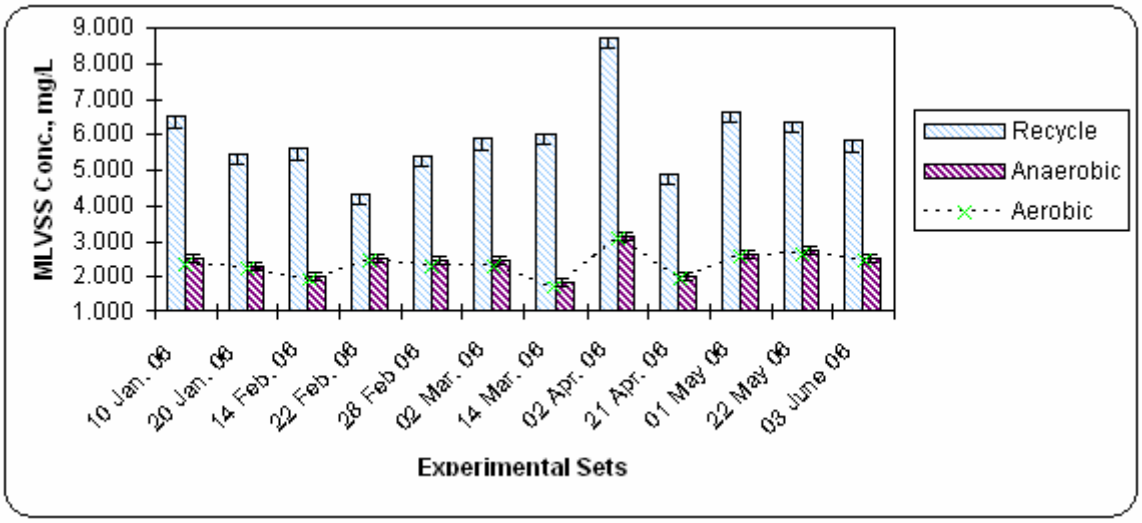


Figure 4.2 MLVSS concentrations in anaerobic, aerobic zones and return sludge

Table 4.1 Chemical Characteristics of Influent

Exp. Set No.	Temp. °C	pH -	DO mg/L	COD mg/L	sCOD mg/L	rbsCOD mg/L	TN mg/L	TP mg/L	PO_4-P mg/L
I(10 Jan. 06)	17	7.9	1.8	780	220	186	38	7.5	5.4
II(20 Jan. 06)	16	8.0	2.5	710	208	182	48	8.1	5.8
III(14 Feb. 06)	15	8.2	2.1	430	154	117	35	6.5	4.5
IV(22 Feb. 06)	16	8.1	2.5	500	185	153	38	9.5	5.4
V(28 Feb. 06)	16	8.2	2.1	440	170	152	35	7.0	4.2
VI(02 Mar. 06)	16	7.9	2.1	420	180	159	40	9.0	5.0
VII(14 Mar. 06)	15	8.3	2.8	290	130	94	32	9.5	6.2
VIII(02 Apr. 06)	19	8.1	2.5	350	160	125	34	8.5	5.1
IX(21 Apr. 06)	21	8.3	1.6	300	112	76	25	5.5	3.9
X(01 May 06)	20	7.6	2.4	380	170	128	36	11.1	5.5
XI(22 May 06)	20	7.6	1.4	740	248	194	40	12.6	9.1
XII(03 June 06)	22	8.2	1.5	370	130	114	37	9.6	5.4
Minimum	15	7.6	1.4	290	112	76	25	5.5	3.9
Average	18	8.0	2.1	476	172	140	37	8.7	5.5
Maximum	22	8.3	2.8	780	248	194	48	12.6	9.1

4.3 Effect of anaerobic retention time on EBPR characteristics

The average rbsCOD/TP ratio was of 16.5. While EBPR efficiency was 55% at 0.84 h AART and with 24.8 rbsCOD/TP ratio, it was 80% at 1.13 h AART with 18.7 rbsCOD/TP ratio. Therefore, most probably AART was limiting the EBPR. In the fact that there was a significant correlation between AART and EBPR efficiency (r^2=0.98). Figure 4.3 indicates that EBPR efficiency was a function of AART and minimum 1.1 hours anaerobic retention time was required to maintain stable EBPR performance.

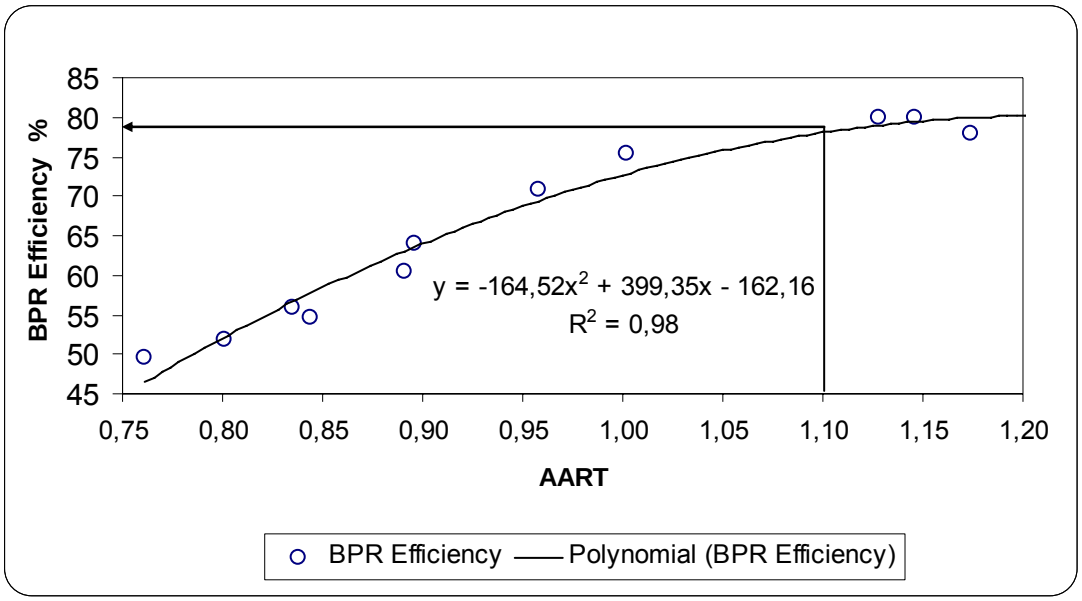


Figure 4.3 Relationship between AART and EBPR efficiency

On the other hand, relationship between the HRT_a and EBPR efficiency (r^2=0.88) was smaller as compared to the AART (r^2=0.98) which is demonstrated in Figure 4.4. These results showed that performance of EBPR processes should be evaluated considering all wastewater characteristics, operational conditions and especially presence of electron acceptors in the anaerobic zone. According to the Figure 4.3, approximately 80% EBPR efficiency could be achieved at 1.1 hours of AART. However according to the Figure 4.4, 80% EBPR efficiency could be affordable for a HRT_a between 1.25 to 1.30 hours. It is evident from these both figures that use of

AART over HRT_a seems more reliable to evaluate the performance of EBPR processes.

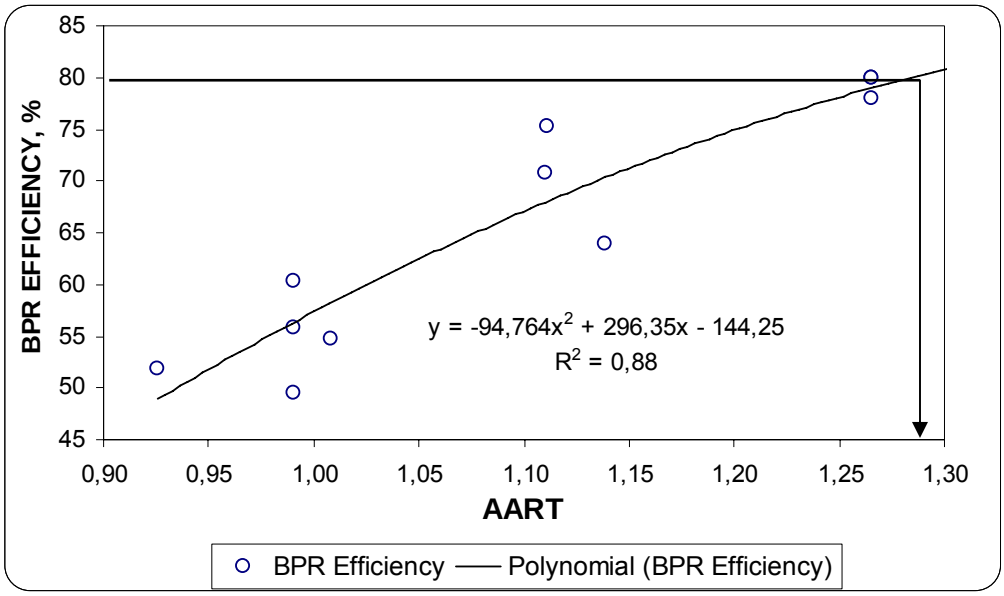


Figure 4.4 Relationship between HRT_a and EBPR efficiency

Another strong indication of a stable EBPR is enrichment of activated sludge microorganisms with an excess amount of P ranging between 4.1 to 15.6% in enhanced mixed-culture conditions based on MLVSS (Mino et al., 1987; Reddy 1991). During the monitoring period, P content of mixed sludge varied from 5.1% to 10.2% with an average P content of 8.0% based on MLVSS. Figure 4.5 demonstrates that P enrichment of activated sludge was also dependent on AART.

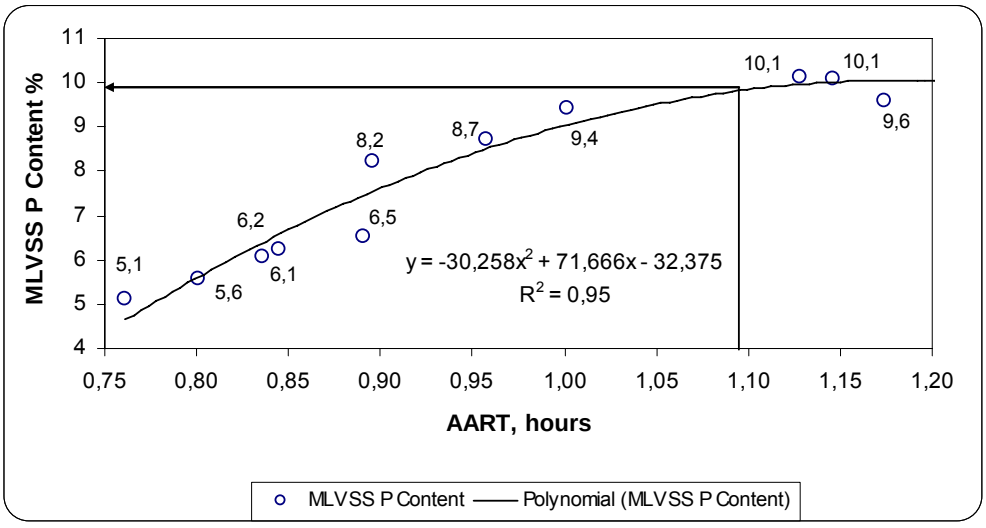


Figure 4.5 P enrichment of activated sludge culture as a function of AART

It could be seen from Figure 4.5 that required AART in anaerobic zone of EBPR process was also confirmed once more by this result as 1.1 hours for ideal conditions that means no electron acceptor input to the anaerobic zone.

According to the previously performed batch experimental results, conducted on the activated sludge obtained from the same WWTP, average P release rate was 0.019 mg P/gVSS min for glucose addition and maximal anaerobic P release rate was 1.63 mg P/g VSS/min as an average for acetate addition. The average P uptake rates were 0.0031 and 0.0046 mg P/gVSS min. It was also reported that the anaerobic sCOD consumption rate was 0.25 mg COD/gVSS min. as an average with acetate addition and it was 0.35 mg COD/gVSS min. as an average with glucose addition (Pala and Bölükbaş, 2005). During monitoring period of this study, measured maximum P release rate was 0.07 mgP/gVSS min (4.2 mgP/gVSS h) and maximum P release rate was 0,006 mgP/gVSS min (0.4 mgP/gVSS h). Therefore measured maximum P release rate was higher than the results after glucose addition but lower than the results with acetate addition. However measured maximum P release rate was higher than the previously obtained maximum P release rates. Plotting measured P release rates versus P release rates, a linear relation was also found with a determination coefficient $r^2 = 0.97$ shown in Figure 4.6.

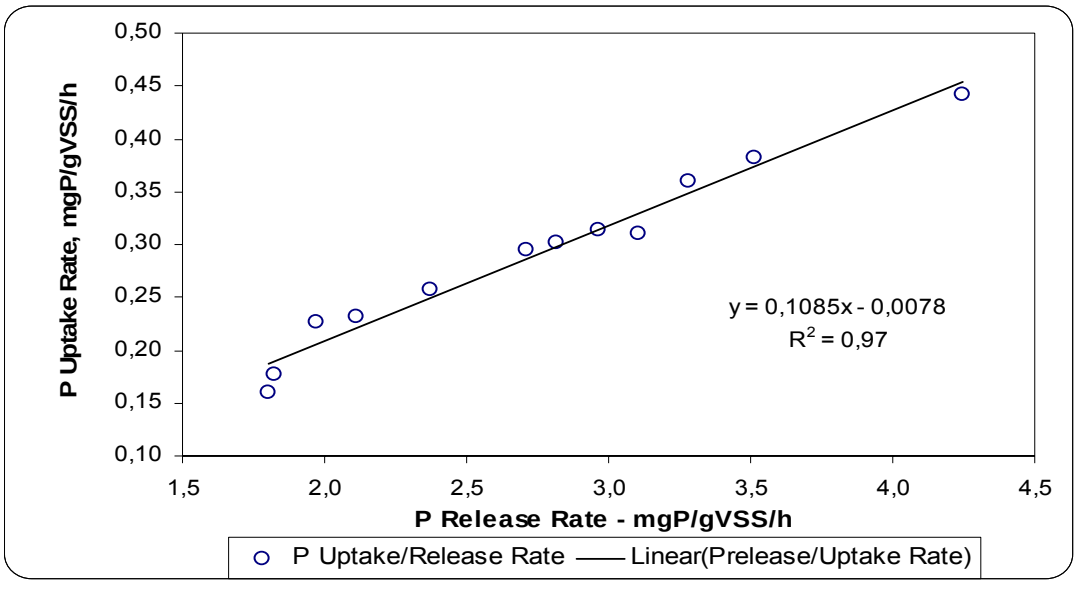


Figure 4.6 Correlation between P_{uptake} and $P_{release}$ Rates

During monitoring period of this study, anaerobic sCOD utilization rate ranged from 0.275 to 0.610 mg sCOD/gVSS/min with an average anaerobic sCOD utilization rate of 0.386 mg sCOD/gVSS/min. Measured sCOD utilization rates were also higher than the reported average sCOD utilization rates.

DO, nitrate and superposed nitrate concentrations entering to the anaerobic zone was demonstrated in Figure 4.7. It should be noted that "superposed nitrate" expresses the total amount of electron acceptors, including nitrate and DO, entering to the anaerobic zone and this figure was obtained from mass-balance equations. In establishing all mass-balance equations, both flow rates and concentrations were used. Measured DO concentration in anaerobic tank was nearly constant at around 0.2 mg/L and this result also indicated that anaerobic tanks nearly reached anoxic conditions. Although the results suggested that as long as the influent soluble C source and HRT_a were within the acceptable limits, a stable EBPR efficiency could be achievable. However, long term operational experience indicates a serious filamentous growth causing the sludge volume index (SVI) to increase up to 200 ml/g, which may create another problem for the effluent quality. One of the main reasons of SVI increase could be explained by presence of electron acceptors in the anaerobic phase (Falkentoft, 2002).

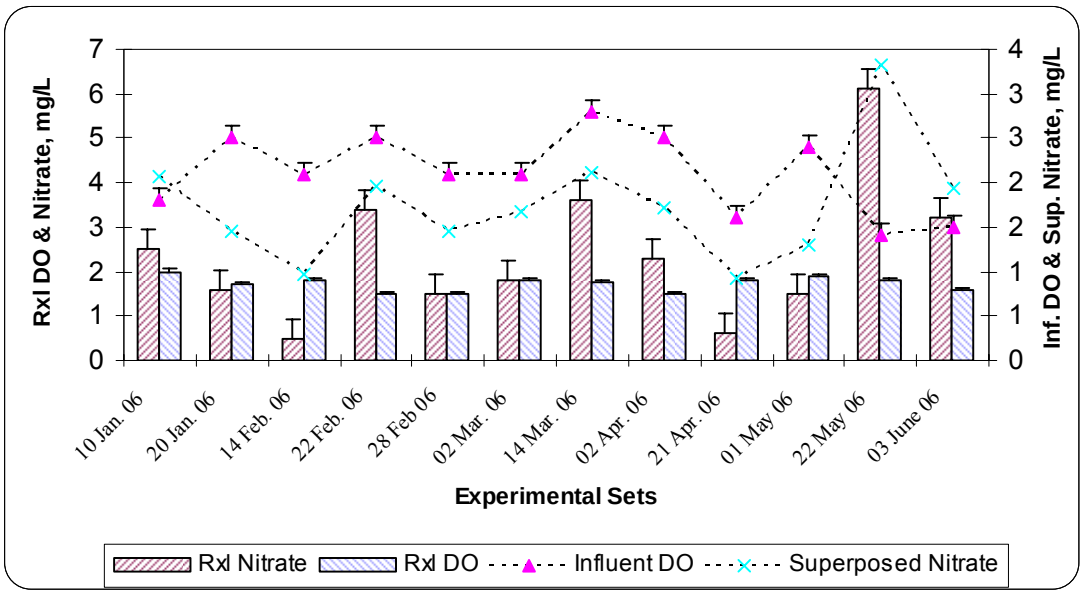


Figure 4.7 Electron acceptor input to the anaerobic zone

4.4 Phosphorus profile and BNR efficiency during the monitoring period

Soluble P profile of the treatment plant is demonstrated in Figure 4.8. It can clearly be seen from these results that the process behavior fitted to typical EBPR mechanism including significant anoxic P removal. It should be noted that the concentration of P removed in anoxic and aerobic zones could only be determined by establishing mass balances for this type process configuration as it was described in the previous chapter. During the monitoring period PO_4-P concentration in aerobic zone ranged from 5.0 to 0.6 mg/L and average PO_4-P concentration was 2.5 mg/L.

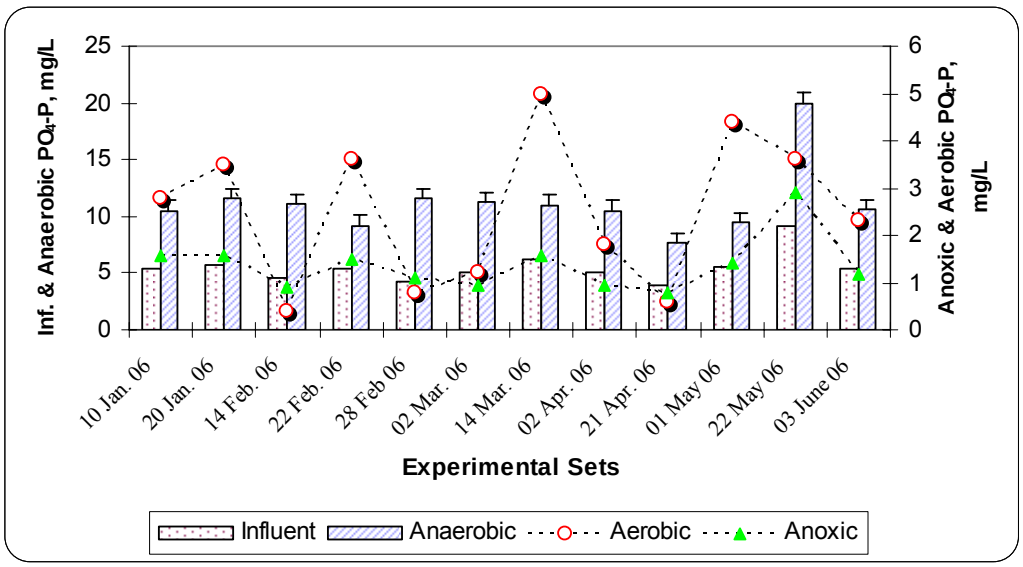


Figure 4.8 PO_4-P profile of the WWTP during the monitoring period.

Average BNR efficiencies were depicted in Figure 4.9. COD removal performance of the described process ranged from 83 to 94% with an average COD removal efficiency of %88. TN removal efficiency varied from 67% to 85% with an average of 78%. EBPR efficiency was unstable and ranged from 49% to 80%. Figure 4.9 also indicates that COD removal was not too sensitive to flow fluctuations. However EBPR was seriously affected by flow variations. TN removal efficiency was not as sensitive as EBPR efficiency but it could be said that it is more sensitive to flow fluctuations than COD removal efficiency. This situation was clearer when the treatment plant efficiency was examined on the basis of 24-h composite sample measurements. Average removal performances of the WWTP, based on 24-h

composites, were 90% for COD, 80% for TN and %50 for P during the monitoring period

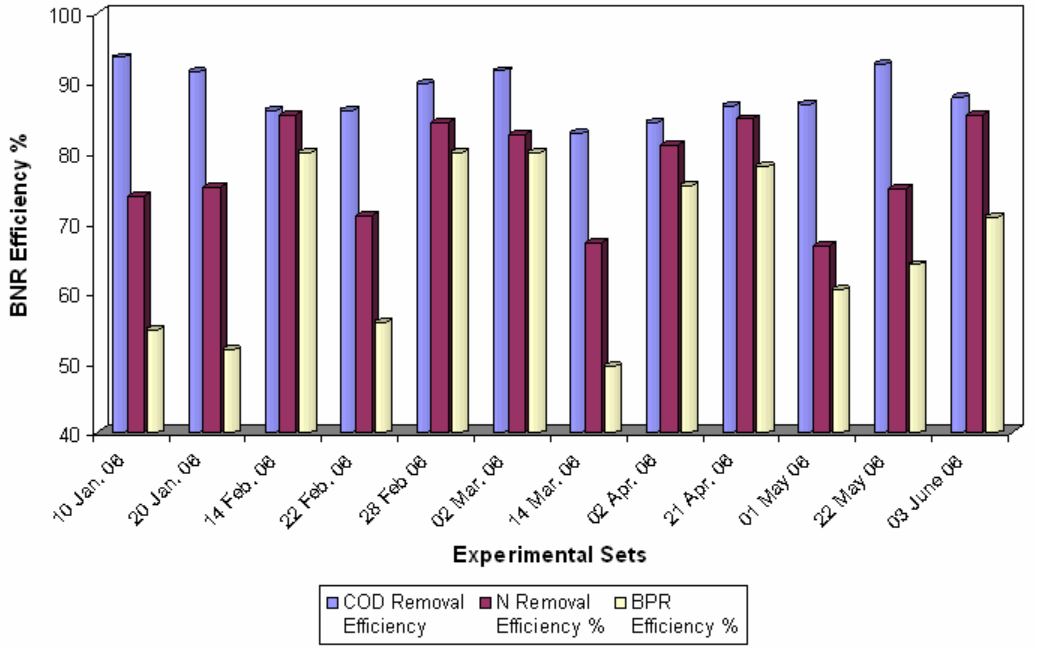


Figure 4.9 BNR Efficiency of the WWTP during the monitoring period.

CHAPTER FIVE
EFFECTS OF SEASONAL rbsCOD/TP VARIATION UPON FUNDAMENTAL CHARACTERISTICS OF FULL-SCALE BIOLOGICAL PHOSPHORUS REMOVAL SYSTEM

5.1 Wastewater characterization and operational conditions

Wastewater composition is one of the critical issue for both design and operation period of BNR plants. Nutrient levels of the wastewater could be used for selection process configuration at design level and could also be used to predict the effluent quality during operation period (Brdjanovic, 1998; Shehab et. al., 1996).

Table 5.1 Group statistics of the seasonal carbonaceous content

Parameter	Season	Data Number	Mean	Std. Dev.	Std. Error Mean
BOD	Winter	24	194	61.398	14.891
	Summer	24	106	14.545	5.498
COD	Winter	24	529	176.902	42.905
	Summer	24	342	40.708	15.386
bCOD	Winter	24	318	100.570	24.392
	Summer	24	174	23.439	8.559
rbsCOD	Winter	24	159	54.979	13.334
	Summer	24	68	10.885	4.414
VFAs	Winter	24	64	21.717	5.267
	Summer	24	31	4.889	1.848
rbsCOD/TP	Winter	24	16	4.169	1.011
	Summer	24	9	2.434	0.920

Investigation results indicated that wastewater organic composition could show seasonal drastic fluctuations. As it could be seen from the statistical analysis results given in Table 5.1 and Table 5.2, influent carbonaceous composition of winter and summer seasons were significantly different from each other. Especially, vital nutrients such as rbsCOD, VFAs and rbsCOD/TP ratio for preferential selection of PAOs within the activated sludge system, decreased significantly in the summer period.

Table 5.2 Statistical comparison of seasonal influent carbonaceous content

t-test results				
BOD_5, COD, bCOD, rbsCOD, VFAs, rbsCOD/TP			95% Confidence Interval of the difference	
Sig. (2-tialed)	Mean Dif.	St. Err. Dif.	Lower	Upper
0.001	88.18	15.874	55.039	121.331
0.001	186.26	45.581	91.031	281.490
0.000	144.60	25.951	90.967	198.809
0.000	90.20	13.955	60.967	119.437
0.000	33.24	5.582	21.577	44.911
0.000	7.16	1.367	4.853	10.578

Average rbsCOD and VFAs concentration of summer period was less than winter period 90.20 mg/L and 33.24 mg/L. Since decrease in carbonaceous content was higher than decrease in P content, average seasonal difference of rbsCOD/TP ratio was 7.16 mg/L. It is also evident from Table 5.2 that, concentration of biodegradable organic materials (BOD_5) was also significantly decreased in the summer period.

Table 5.3 Descriptive statistics of the influent N, P content

Parameter	Data Groups	Data Number	Mean	Std. Dev.	Std. Error Mean
TN	Winter	24	36.7	4.820	1.169
mg/L	Summer	24	27.8	4.486	1.695
NH_4-N	Winter	24	23.0	5.540	1.512
mg/L	Summer	24	23.5	4.750	1.942
TP	Winter	24	9.4	2.490	0.605
mg/L	Summer	24	7.6	0.811	0.306

As it could be seen from the Table 5.3 and Table 5.4, variation in TN, NH_4-N and TP were not critical as compared to the decrease in carbonaceous level. Especially seasonal NH_4 - N concentration of the influent was nearly the same (2-tailed sig.:0.835). Average difference of influent TP concentration between the periods was 1.836 mg/L. Since the decrease of carbonaceous content was much higher as

compared to decrease in TP concentration, average summer rbsCOD/TP ratio was significantly lower than winter period up to 56%.

Table 5.4 Statistical comparison of influent N, P content

(TN, NH_4-N, TP)			95% Confidence Interval of the difference	
Sig. (2-tialed)	Mean Dif.	St. Err. Dif.	Lower	Upper
0.000	8.970	2.124	4.564	13.377
0.835	-0.525	2.461	-5.932	4.881
0.073	1.836	0.679	0.425	3.246

Environmental factors including pH and temperature were within the acceptable range for EBPR to occur (Brdjanovic, 1997; 1998). It could be seen from Figure 5.1 that maintained microorganism concentrations in the process were at sufficient level. This figure also indicated that sludge age was also good enough to obtain adequate level of biological treatment. Previously studies showed that sludge age is the one of the critical parameters for EBPR in combination with temperature. It was reported that optimum sludge age for adequate EBPR efficiency should be determined considering the temperature (Erdal et al., 2003; Brdjanovic et al., 1998). During the monitoring period sludge age ranged between 12 (at 19.4 °C) and 18 (at 19.7 °C) days with an average value of 15 days. It could be concluded that maintained sludge age level during monitoring period were good enough and most probably did not limit the EBPR process.

Another important operational parameter in EBPR is HRT_a. Although the optimum HRT_a varies according to the process type, it could be said that HRT_a should be higher than 1 hour for many EBPR processes (WPCF, 1991; USEPA 1987; Metcalf and Eddy 2003).

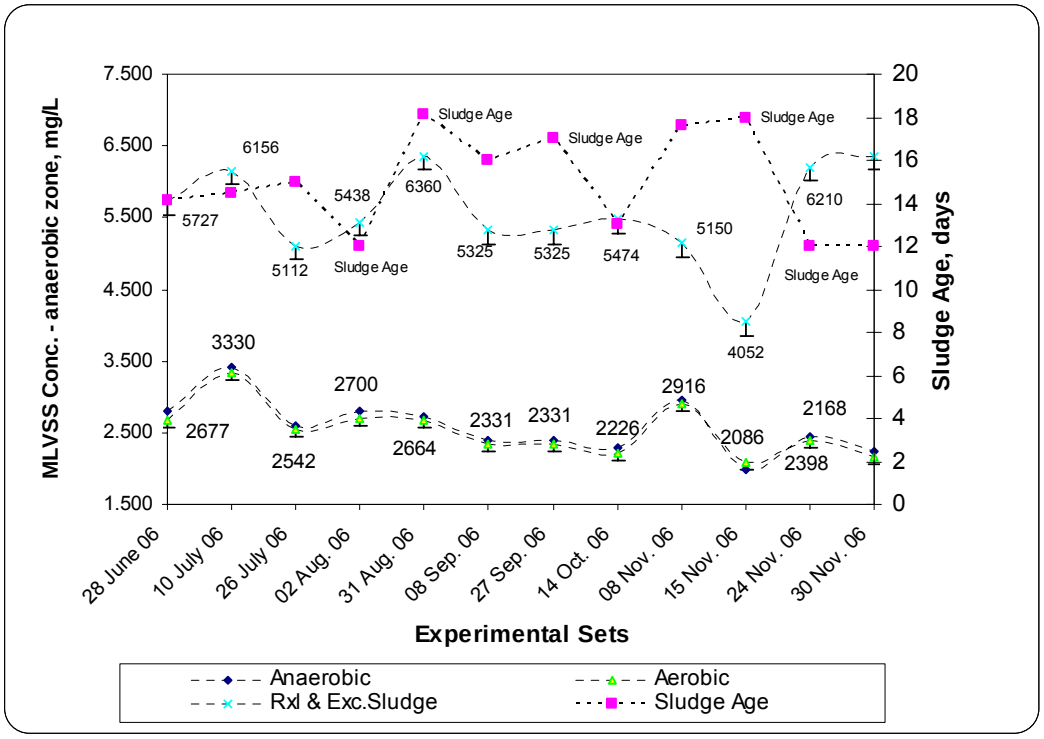


Figure 5.1 MLVSS concentrations at various stages of the process and sludge age

In this study, to obtain comparable results and evaluate the EBPR process accurately, samples were taken at similar flow rates, corresponding to nearly equal HRT_a values of higher than 1 hour. Estimated HRT_a values were depicted in Figure 5.1

5.2 Effects of rbsCOD/TP variation on EBPR characteristics

Many scientific investigations showed that performance of EBPR processes highly depends on influent COD/TP ratio and EBPR efficiency improves with increasing COD/P ratio (Park et al., 1997; Abu-ghararah et al., 1991; Ekama and Marais, 1984). Parallel to these studies, it was found that rbsCOD/TP ratio interacts with many fundamental characteristic of the EBPR process. It could be seen from Figure 5.2 that seasonal variation of rbsCOD/TP ratios seriously effected EBPR performance. Long term operational experiences showed that organic composition of influent decreases during the summer period and increases in winter period. The influent is composed of domestic and pre treated industrial wastewater.

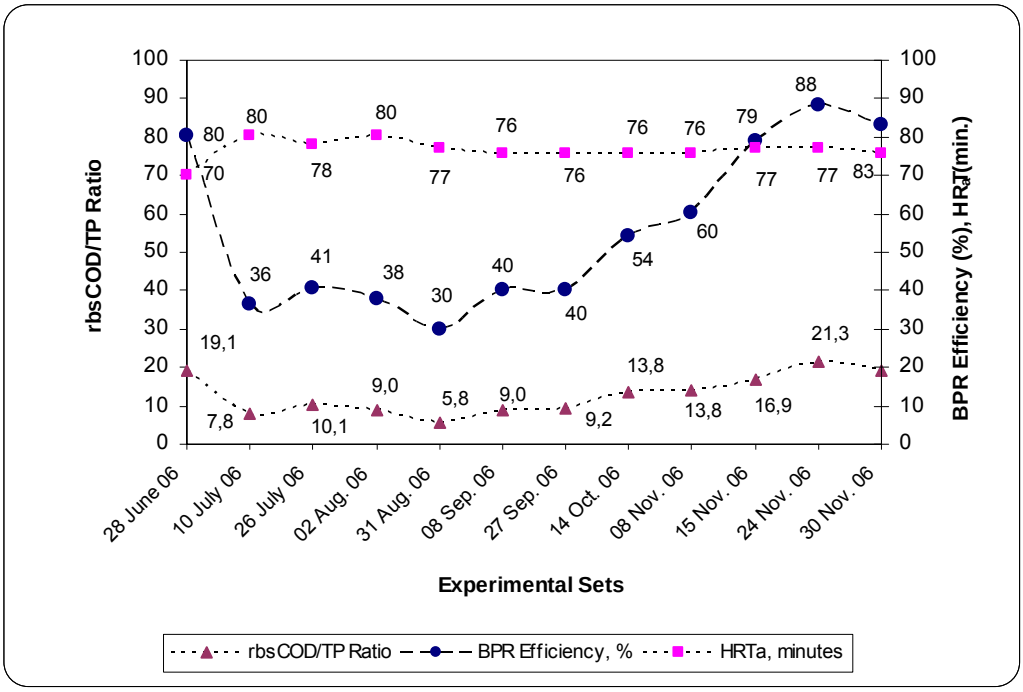


Figure 5.2 Recorded rbsCOD/TP ratio, EBPR efficiency and HRT_a values during the monitoring period.

Furthermore, domestic portion of the influent could be expected to decrease in summer period due to population move to summer houses. As a result of decrease in domestic wastewater percent, rbsCOD /TP ratio of the influent could be expected to decrease in summer season as a result of increasing pre treated industrial portion. Seasonal decrease of rbsCOD/TP ratio and deterioration of EBPR efficiency was depicted in Figure 3. It could be said that rbsCOD/TP ratios lower than 10 resulted with only metabolic P removal that occurs through the microbial growth (30% EBPR efficiency) without excess EBPR removal.

Relationship between EBPR efficiency and rbsCOD/TP ratio also demonstrated in Figure 5.3. This figure indicates that there were a linear correlation between these variables with a determination coefficient of r^2=0.98. However, plotting COD/TP ratios versus EBPR efficiencies, a linear relationship was found with a determination coefficient of r^2=0.62 as it was depicted in Figure 5.4.

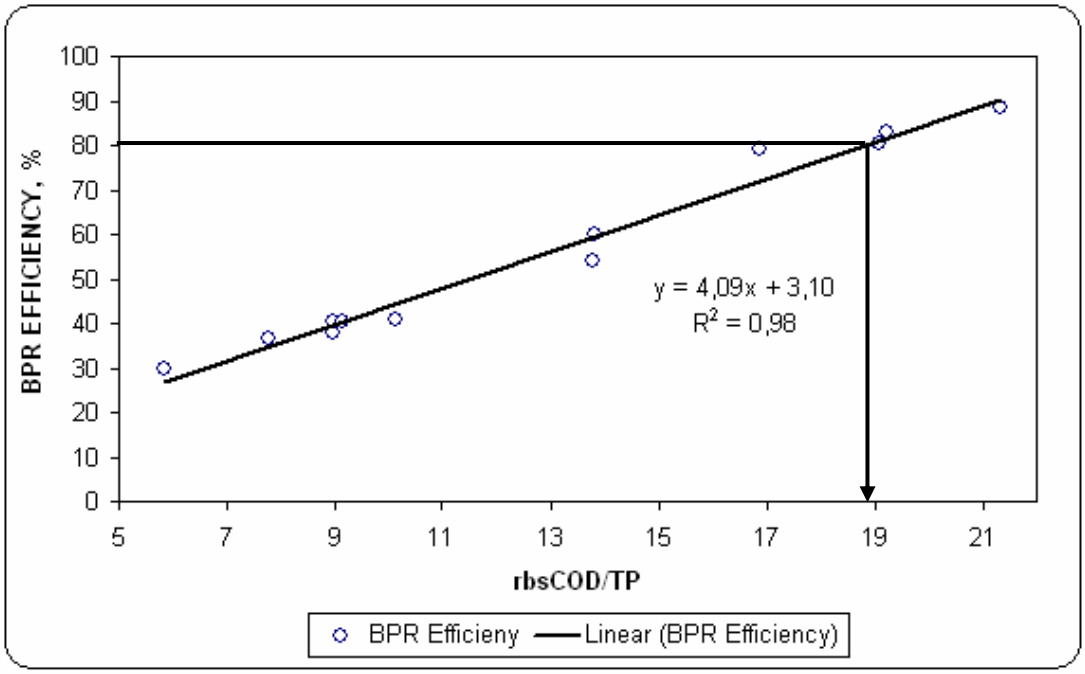


Figure 5.3 Relationship between rbsCOD/TP ratio and EBPR efficiency.

According to the Figure 5.4, an adequate EBPR efficiency (80%) could be obtained at COD/TP ratio of higher than 65. This ratio was higher than the previously reported values (Randall et al. 1992).

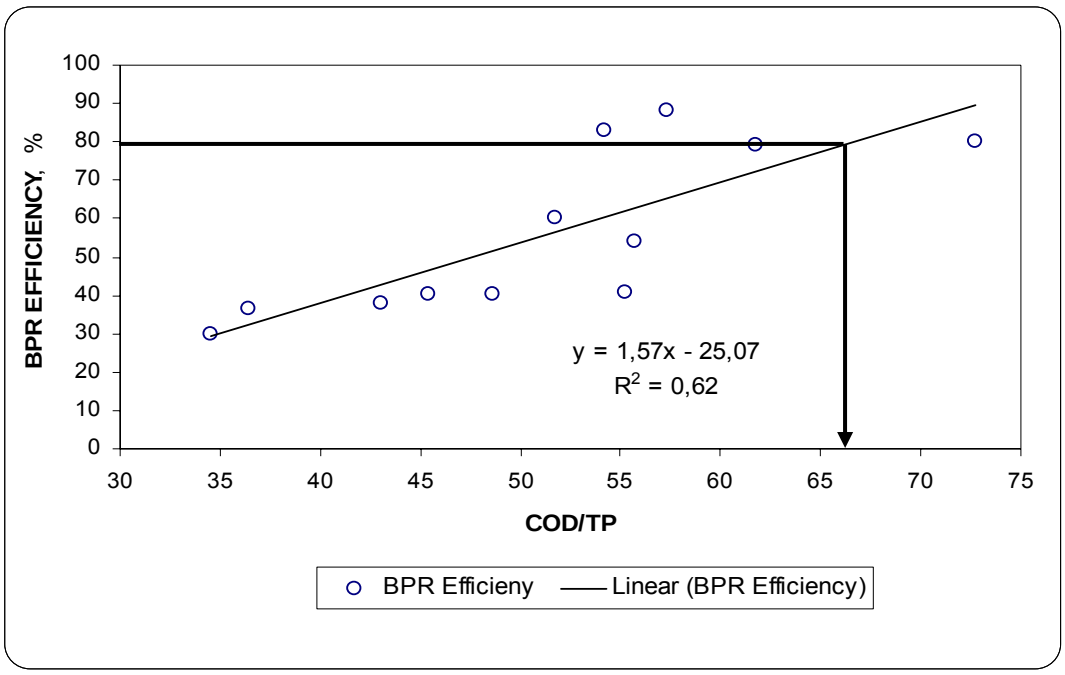


Figure 5.4 Relationship between COD/TP ratio and EBPR efficiency.

However 80% - EBPR efficiency was achieved at rbsCOD/TP ratio of nearly 19 similar to previously findings as it was shown in Figure 5.3. Therefore use of rbsCOD/TP ratio over COD/TP ratio seems more accurate and reliable to evaluate the performance of full-scale EBPR processes. In the fact that the fractions of COD mainly depend on the sewerage network length. Furthermore the portion of sCOD increases with increasing length of sewerage network line. Another advantage in use of rbsCOD/TP ratio as a performance criterion is simplicity of the required experimental procedures for WWTPs.

Investigations indicated that rbsCOD/TP ratio not only affected the performance of the system but also affected many fundamental characteristics of EBPR process. These characteristics mainly composed of enrichment of activated sludge with excess amount P, anaerobic COD utilization, P release and uptake rates.

Enrichment of activated sludge microorganisms with excess amount of P ranging from 4.1 to 15.6% based on MLVSS is an important indicator for successful EBPR process (Mino et al., 1998; Reddy, 1991). This parameter could also be an indication of PAOs domination in the activated sludge system.

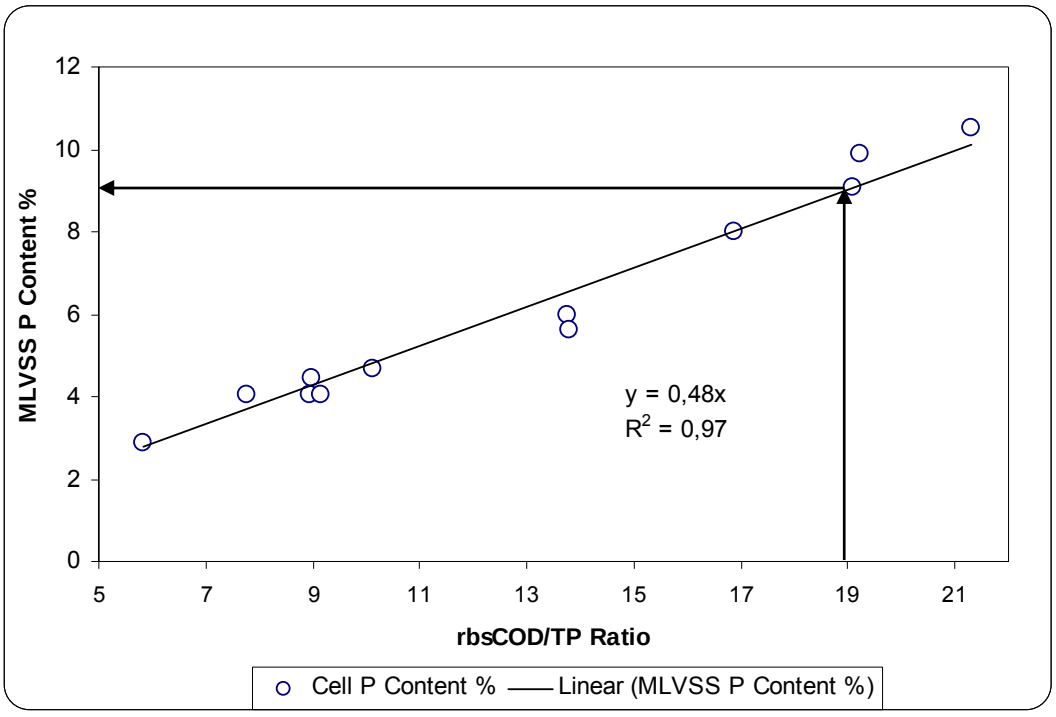


Figure 5.5 Relationship between rbsCOD/TP ratio and P content of MLVSS

During monitoring period of the study, P content of mixed sludge varied from 2.9% (at rbsCOD/TP:6) to 10.5% (at rbsCOD/TP:21) with an average P content as 6.1% based on MLVSS. Figure 5.5 shows that there was a linear relationship between rbsCOD/TP ratio and P content of MLVSS with a determination coefficient of r^2=0.97. it is also evident from figure 5.5 that adequate bacterial phosphorus enrichment could be achieved at rbsCOD/TP ratio of 19.

EBPR process could mainly be characterized with consumption of readily biodegradable soluble substrates (rbsCOD) and release of P into liquid interface in the anaerobic zone and the uptake of excess amount of P into microbial cell in the following aerobic zone. Previous investigation results proposed that there were significant correlation between P uptake and P release ratio ($P_{uptake}/P_{release}$) ranging from 1.15 to 1.20 (Park et al., 2001). In this study, $P_{uptake}/P_{release}$ ratios varied from 1.12 (at rbsCOD/TP:6) to 1.22 (at rbsCOD/TP:21).

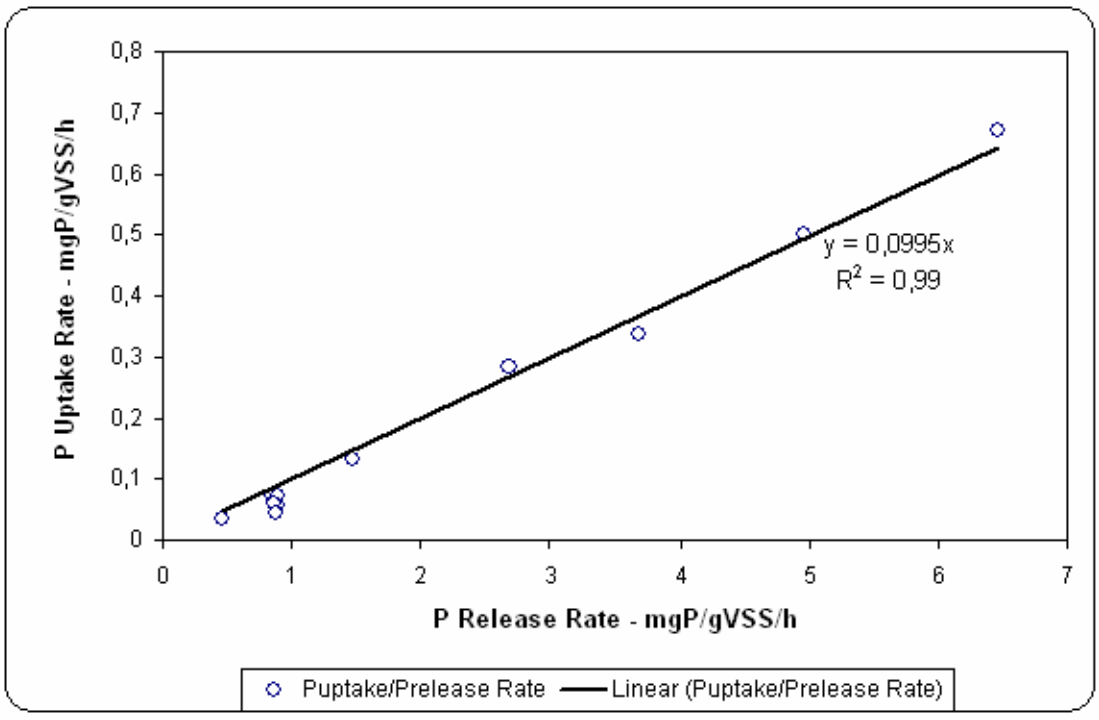


Figure 5.6 Correlation between P Uptake and Release Rates

Other identical parameters of EBPR are P release and P uptake rates. Average anaerobic P release rate, measured by laboratory-scale batch test was reported as 1.63 (Pala and Bölükbaş, 2005) and 0.1 (Brdjanovic et al. 1998) mgP/gVSS/min. These results were obtained using acetate as carbon source.

In this study, maximum anaerobic P release rate was 0.108 mgP/gVSS/min (at rbsCOD/TP:19) and minimum P release rate was 0.008 mgP/gVSS/min (at rbsCOD/TP:6). Average aerobic P uptake rate was reported as 0.0031 (Pala and Bölükbaş, 2005) and 0,037 (Brdjanovic et al., 1998) mgP/gVSS/min. During the monitoring period of this study, measured maximum P uptake rate was 0.011 (at rbsCOD/TP:19) and measured minimum P uptake rate was 0.001 (at rbsCOD/TP:6). It could be concluded that increasing influent rbsCOD/TP ratio resulted with higher P release and P uptake rates.

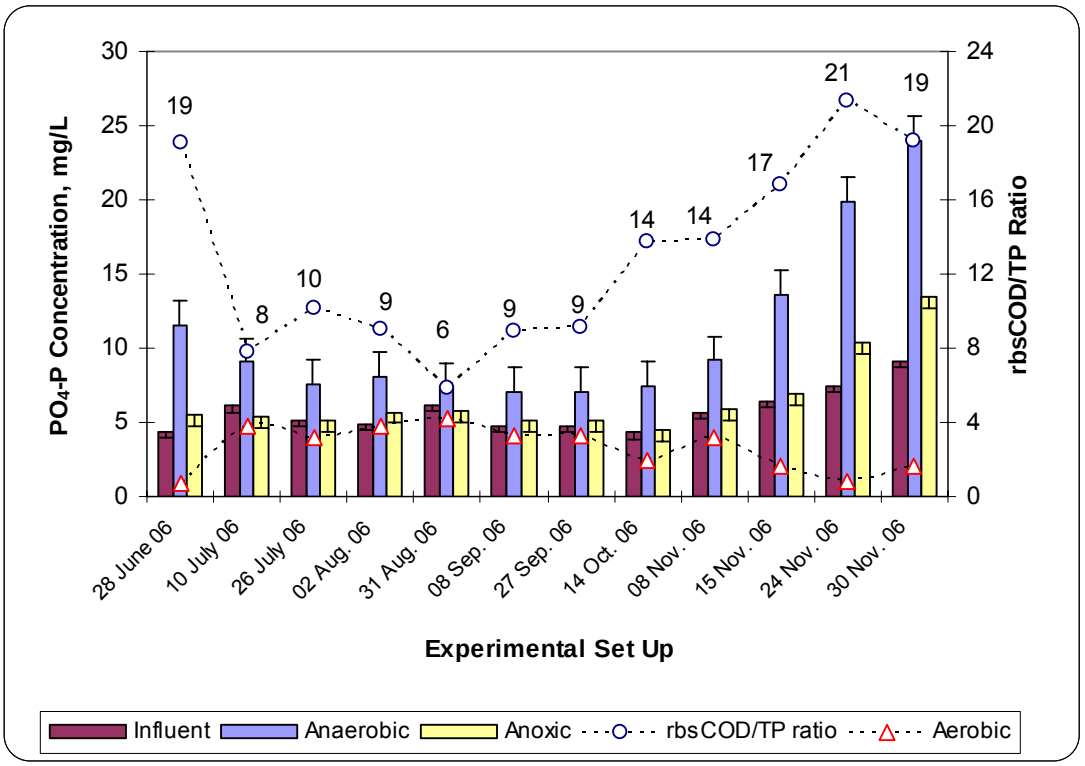


Figure 5.7 Measured PO_4-P concentration in the process and rbsCOD/TP ratio during the monitoring period

Ability of microorganisms to utilize organic matter mainly in the form of VFAs in the anaerobic period and simultaneous release of P into liquid interface is another

important indication of successful EBPR process. In these biochemical reactions, required energy for COD (VFAs) utilization supplied with release of P in the form of PO_4-P. COD utilization rate in anaerobic zone was reported as 0.25 mgCOD/gVSS/min for acetate addition and 0.35 mgCOD/gVSS/min for glucose addition (Pala and Bölükbaş, 2005). In this study, measured maximum COD utilization rate was 0.61 mgsCOD/gVSS/min (at rbsCOD/TP of 19 and EBPR efficiency of 83%) and minimum COD utilization rate was 0.05 mgsCOD/gVSS/min (at rbsCOD/TP of 6 and EBPR efficiency of 30%). Summary of the obtained results and comparison with previously findings are given in Table 5.5 and it could be concluded that the ability of activated sludge to utilize COD by the aid of P release in anaerobic period could depends on rbsCOD/TP of influent. As it could be seen from Figure 5.7, the amount of P released in anaerobic period was proportional to influent rbsCOD/TP ratio. It should be noted that all data used in the correlation graphics were not in the time order. These data series are a mixture of measured values during various periods of the monitoring period.

Table 5.5 Comparison of the important EBPR characteristics with previously obtained results

Reference	**$P_{Release}$ Rate** mgP/gVSS/min	**P_{Uptake} Rate** mgP/gVSS/min	**$P_{uptake}/P_{release}$ Ratio**	**sCOD Util. Rate** mgsCOD/gVSS/min
Brdjanovic et al., 1998	0.10	0.037	-	-
Pala and Bölükbaş, 2001	1.63	0.003	-	0.25
Park et al. 2001	-	-	1.15-1.20	-
In this study (maximum)[a]	0.11	0.011	1.12	0.61
In this study (minimum)[b]	0.01	0.001	1.22	0.05

[a]rbsCOD/TP ratio ≥ 19; [b]rbsCOD/TP ratio = 6

CHAPTER SIX
ACTIVATED SLUDGE CHARACTERIZATION

In this chapter previously obtained full scale measurement results and the established mass balances were used for characterization of the activated sludge community in terms of PAOs and non-PAOs cell synthesis and phosphorus enrichment of the microorganisms. Actually it is possible to measure the P content of the activated sludge by analytical techniques as described in the Chapter III. However P content of the PAOs could not be measured directly without additional techniques. In this part of the thesis using the previously described approaches and assumptions given in the Chapter III, P content of the activated sludge and PAOs were estimated. Since activated sludge P content was determined both using analytically technique and mass balance equations, accuracy of the used approach to predict PAOs cell synthesis and P enrichment were comparable.

Most of the heterotrophic bacteria in the activated sludge system, typically contains %1.5-2 intracellular P. However many bacteria have the ability of P storage as high as 20 to 30 percent based on dry weight. According to the theory of EBPR, rbsCOD utilization resulting with acetate uptake is very important in determining the produced amount of PAOs. If significant amount of electron acceptors input to the anaerobic zone, the acetate could be depleted before it is taken up by PAOs. Since rbsCOD could be converted to acetate easily in shot anaerobic retention times, influent rbsCOD level is very important in determining the quantity of P removed by the biological storage. Investigations showed that low influent rbsCOD concentrations rapidly lead to deterioration in EBPR efficiency (Metcalf and Eddy, 2003)

Identification of activated sludge with particular reference to EBPR indicated that heterotrophic bacteria such as *Pseudomonas*, *Aeromonas* and *Acinetobacter* accomplished denitrification and also these bacteria have the ability of P accumulation. These results showed that polyphosphate accumulation and denitrification in activated sludge could be carried out by the same microorganisms.

It has been reported that *Pseudomonas* and *Acinetobacter* in an anoxic stage coexisted at 55% and 16%, respectively. Also, it has been suggested *Pseudomonas* are actively take part in simultaneous denitrification and phosphorus uptake in EEBPR sludge (Lötter & Murphy, 1985; Jorgensen and Pauli, 1995; Atkinson *et al*., 2001).

A recent laboratory scaled scientific investigation indicated that intracellular P content of PAOs ranged from 24% to 37% for different P: COD ratios and mass fraction of PAOs in the activated sludge culture ranged from 10% to 71%. It was also reported that mass fraction of ordinary heterotrophic microorganisms (OHOs) ranged between 2% to 3% (Panswad et. al., 2007).

6.1 Characterization of the activated sludge exposing to variable HRT_a

To establish required mass balances, detailed characterization of the influent and effluent were performed. Similar wastewater characterization were reported in the Chapter 4 and in addition to these results, biodegradable COD (bCOD) and particulate biodegradable COD (pbCOD) that were determined using the BOD data were reported as well. This additional data were used to estimate non-PAOs cell synthesis and to establish required mass balances. sCOD utilized in the anaerobic period resulting with P release were used to estimate total PAOs cell synthesis. In principle, anoxic PAOs cell synthesis was based on anoxic PO_4-P and NO_3-N utilization that were estimated using mass balance equations as described in Chapter III. Aerobic PAOs cell synthesis was based on the difference between total PAOs growth and anoxic PAOs growth.

Wastewater characterization results indicated that average bCOD concentration of the settled wastewater was 280 mg/L and it could be also said that percentages of the particulate and soluble biodegradable materials is approximately equal. Very long sever network structure could explain the reason of nearly equal particulate and soluble substrate forms. From this initial wastewater characterization it could be expected to have nearly equal quantities of PAOs and non-PAOs in the process.

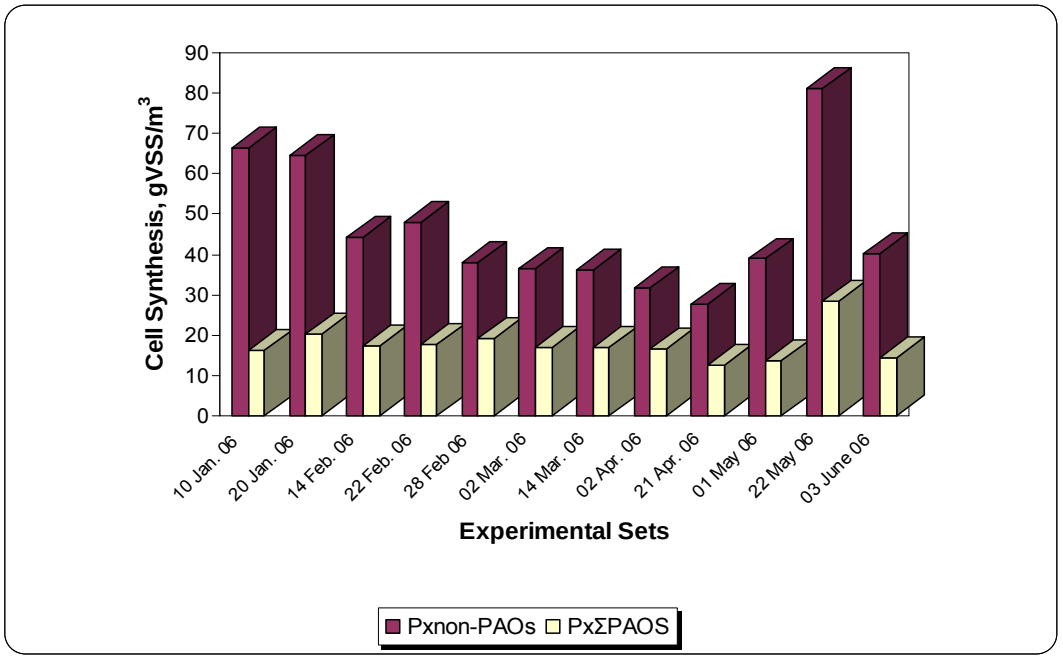


Figure 6.1 Estimated cell synthesis levels during the monitoring period

However it is evident from Figure 6.1 that as a result of variable (insufficient) HRT_a, presence of electron acceptors (detailed investigation results were given in the Chapter 4) in the anaerobic zone and higher growth rate of the ordinary heterotrophic bacteria, level of the cell synthesis and percentage of non-PAOs microorganisms in the activated sludge were much higher than PAOs. Percentages of the microbial groups as a result of above described cell synthesis in which PAOs domination is at minimum and maximum level are demonstrated graphically in Figure 6.2a and Figure 6.2b. As it could be seen from these figures, increase of PAOs percentage from 19% to 34% lead to a EBPR efficiency increase from 55% to 75%. All cell synthesis and P enrichment of both activated sludge and microbial groups recorded during the first monitoring period are given in Table 6.1.

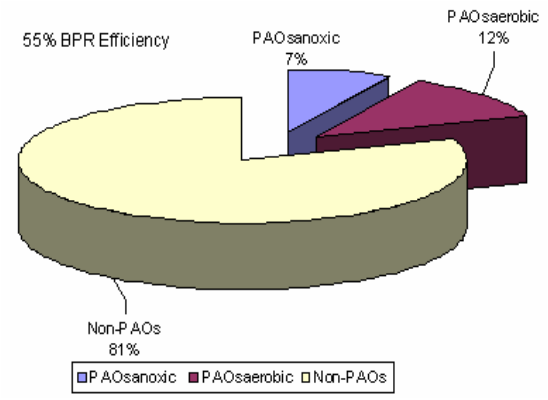


Figure 6.2a Microbial dispersal at minimum PAOs domination

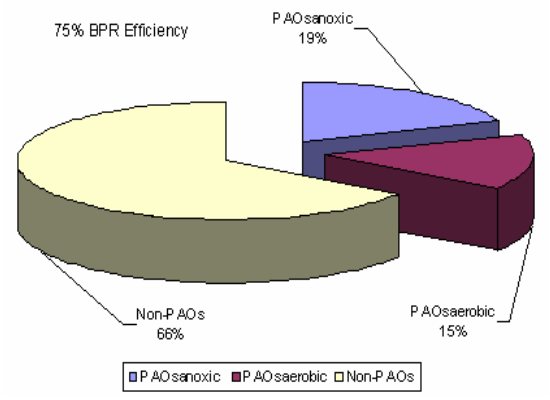


Figure 6.2b Microbial dispersal at maximum PAOs domination

Table 6.1 Cell synthesis and P enrichment of the microbial groups during the monitoring period

$Px_{PAOs\text{-}anoxic}$	$Fp_{PAOs\text{-}anoxic}$	$Px_{non\text{-}PAOs}$	$Fp_{non\text{-}PAOs}$	$Px_{\Sigma PAOS}$	$Px_{PAOs\text{-}aerobic}$	$Fp_{PAOs\text{-}aerobic}$	Fp_{MLVSS}[a]	Fp_{MLVSS}[b]
gVSS/m^3	%	gVSS/m^3	%	gVSS/m^3	gVSS/m^3	%	%	%
6,1	30	66,3	1,8	16,1	10,0	24	6,3	6,2
11,0	27	64,5	1,8	20,3	9,3	18	5,9	5,6
8,8	31	44,1	1,9	17,2	8,4	29	9,5	10,2
8,4	11	47,9	2,0	17,7	9,2	18	6,3	6,1
8,1	30	37,8	1,8	19,1	11,0	27	10,3	10,1
9,7	30	36,5	2,0	16,9	7,2	29	10,5	10,1
9,0	24	36,1	2,0	16,8	7,8	11	4,9	5,1
9,4	32	31,8	1,9	16,7	7,3	21	8,7	9,4
4,5	30	27,6	2,1	12,6	8,1	27	9,7	9,6
10,3	17	38,9	1,9	13,6	3,3	22	7,1	6,5
4,4	29	81,0	1,9	28,3	23,9	29	8,9	8,2
8,3	29	40,1	2,0	14,3	5,9	30	9,4	8,7

Fp_{MLVSS}[a]: Estimated activated sludge P content; Fp_{MLVSS}[b]; Measured activated sludge P content (Px: Cell synthesis; Fp: P fraction of the cell)

Estimated (Fp_{MLVSS}^{a}) and measured (Fp_{MLVSS}^{b}) activated sludge P content at the end of the aerobic period were analyzed using Independent – Samples T Test for 0,95 confidence interval using SPSS v.13 software. According to the group statistics, standard deviation of Fp_{MLVSS}^{a} was 1.91 and Fp_{MLVSS}^{b} was 1.95. Performing T Test, 2-tailed significance value was calculated as 0,859 with a mean difference of 0.141. From these statistical analysis results it could be concluded that measured and estimated activated sludge P content values are very similar. Since the developed approach to determine the PAOs P content directly related to the activated sludge P content, accuracy of the obtained results could be evaluated as acceptable.

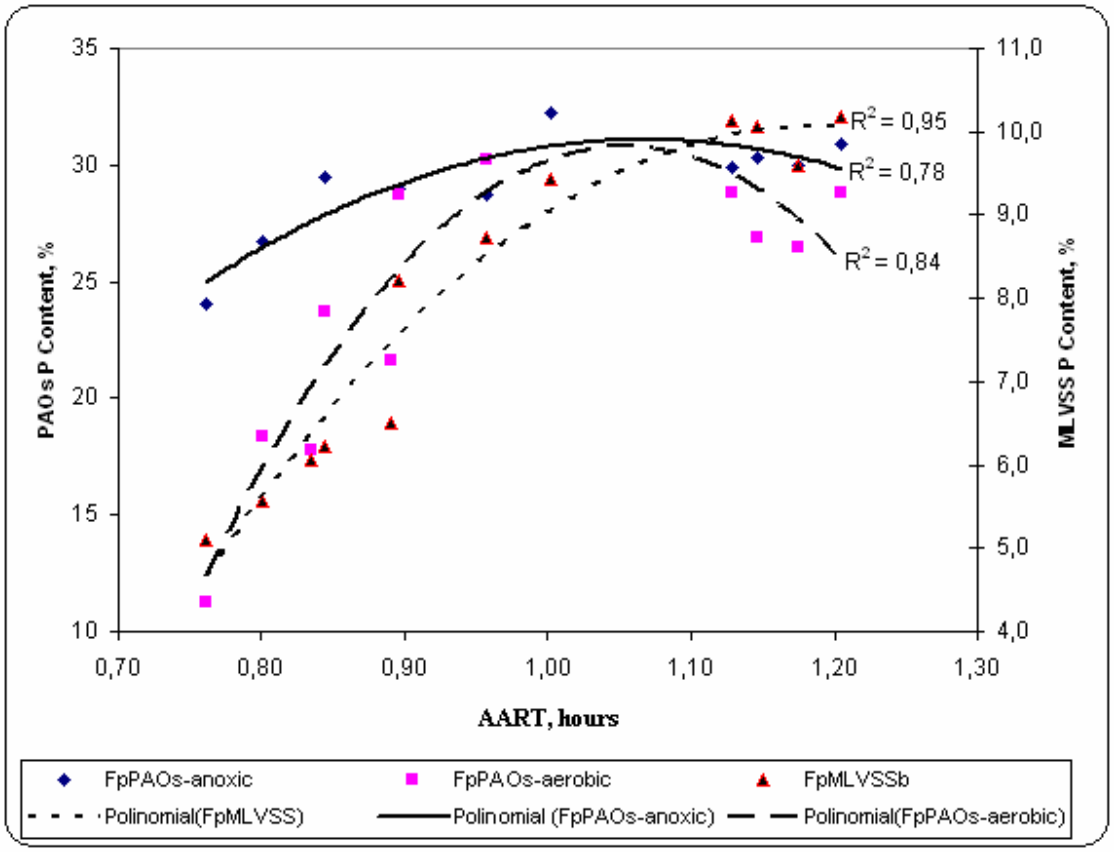


Figure 6.3 Effect of AART on PAOs and activated sludge P content

As it could be seen from Figure 6.3, AART significantly effected internally stored poly-P concentration of different PAOs groups and the activated sludge P content. This situation could be explained by short anaerobic period leading insufficient acetate uptake resulting with poor internal PHB synthesis. Since the internally stored PHB level is low, in the following anoxic and aerobic zones, PAOs could not

generate the required energy to restore Poly-P reserves up to maximum level. Experiments conducted under different HRT_a levels indicated that internally stored P content ranged from 24% to 32% for anoxic PAOs and 11% to 30% for aerobic PAOs. Correlation coefficients were r^2=0.78 for PAOs-anoxic and r^2=0.84 for PAOs-aerobic. Stability of the anoxic PAOs intracellular P level could be explained with their ability to use either nitrate or DO as electron acceptor. However aerobic PAOs could only use DO as electron acceptor as result of that they are active only in strict aerobic conditions.

6.2 Characterization of the activated sludge loading under variable rbsCOD/TP ratios

In this section of the thesis, activated sludge was characterized using the experimental results obtained by full-scale experiments similar to previously section (6.1). As discussed in the Chapter 5 deeply, it was found that seasonal variation of rbsCOD/TP ratio effected many important characteristics of the activated sludge. Therefore, in this part of the thesis activated sludge loading under variable rbsCOD/TP ratios were analyzed to identify the microbial community within the frame of EBPR.

During the monitoring period, rbsCOD varied from 49 mg/l to 275 mg/L with an average of 126 mg/L. pbCOD ranged between 80 mg/L and 225 mg/L and average pbCOD concentration was 141 mg/L in the influent. Similar to the experimental results given in the section 6.1, nearly 50% of the biodegradable COD (bCOD) was in the soluble form and the remaining 50% is in the suspended form. Therefore, it could be said that fractions of COD are mainly related to the sewer network length.

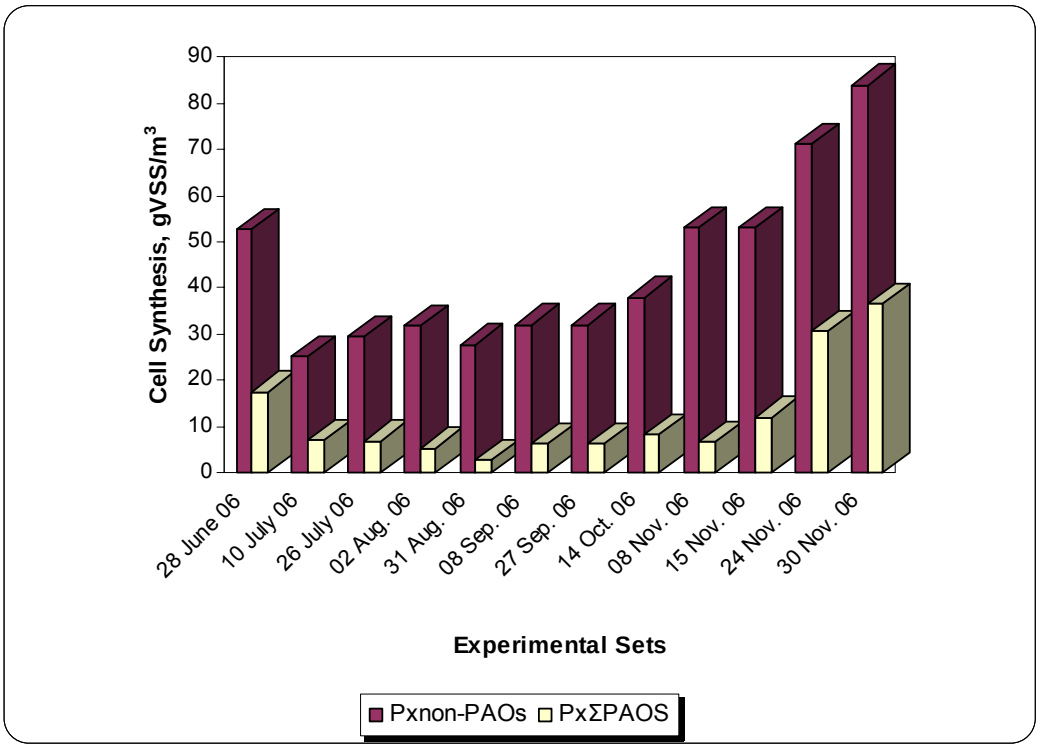


Figure 6.4 Estimated cell synthesis levels during the monitoring period

Increase of the influent bCOD concentration from 146 mg/L up to 498 mg/L with parallel increase of rbsCOD from 66 mg/L up to 275 mg/L gave a boost to both non-PAOs and PAOs bacterial cell synthesis as shown in Figure 6.4. PAOs cell synthesis was increased from 3 gVSS/m^3 to 37 gVSS/m^3 parallel to increase of rbsCOD/TP ratio (depicted in Figure 5.2) from 6 to 21. As a consequence of increasing PAOs cell synthesis, the percentage of these microorganisms also increased from 10% to 30% within the activated sludge system.

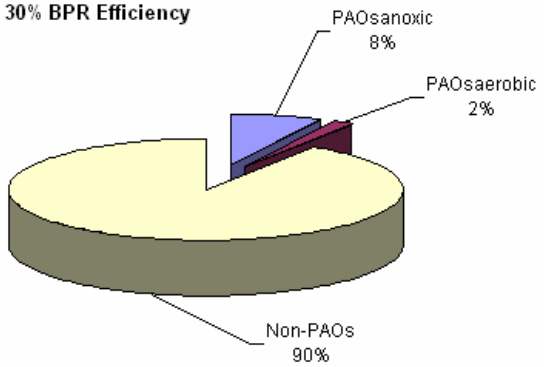


Figure 6.5a Microbial dispersal at minimum EBPR efficiency

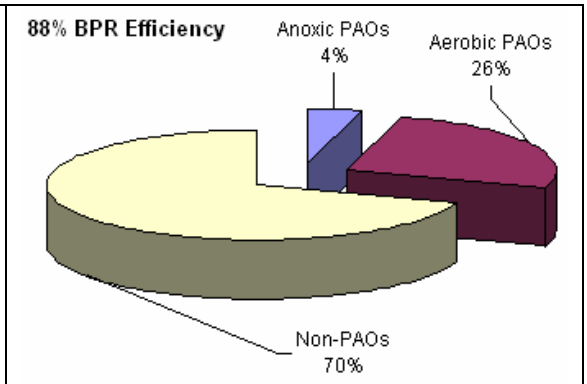


Figure 6.5b Microbial dispersal at maximum EBPR efficiency

Table 6.3 Cell synthesis and P enrichment of the microbial groups during the monitoring period (COD limited conditions)

$Px_{PAOs\text{-}anoxic}$	$Fp_{PAOs\text{-}anoxic}$	$Px_{non\text{-}PAOs}$	$Fp_{non\text{-}PAOs}$	$Px_{\Sigma PAOS}$	$Px_{PAOs\text{-}aerobic}$	$Fp_{PAOs\text{-}aerobic}$	Fp_{MLVSS}[a]	Fp_{MLVSS}[b]
gVSS/m^3	%	gVSS/m^3	%	gVSS/m^3	gVSS/m^3	%	%	%
5,6	30	52,6	1,6	17,5	11,8	30	9,0	9,1
5,7	30	25,1	1,6	7,0	1,2	16	5,1	4,1
3,4	28	29,4	1,9	6,8	3,5	15	4,4	4,7
3,3	27	31,9	2,0	5,3	2,0	18	4,3	4,5
2,3	28	27,6	1,6	2,8	0,5	10	2,7	2,9
3,7	19	32,1	1,7	6,3	2,6	14	3,9	4,0
3,7	19	32,1	2,0	6,3	2,6	14	3,9	4,0
2,6	31	37,9	2,0	8,2	5,5	24	5,9	6,0
4,0	27	53,1	1,7	6,7	2,8	30	5,2	5,6
4,3	18	53,1	2,1	11,9	7,7	24	6,0	5,2
4,2	31	71,1	1,7	30,5	26,3	30	10,5	10,5
3,9	30	83,9	1,7	36,6	32,7	29	10,3	9,9

Fp_{MLVSS}[a]: Estimated activated sludge P content; Fp_{MLVSS}[b]; Measured activated sludge P content (Px: Cell synthesis; Fp: P fraction of the cell)

Table 6.3 presents the estimated cell synthesis levels, P fraction of both PAOs and the activated sludge. Estimated (Fp_{MLVSS}^{a}) and measured (Fp_{MLVSS}^{b}) activated sludge P content at the end of the aerobic period were analyzed using Independent – Samples T Test for 0,95 confidence interval using SPSS v.13 software. According to the group statistics, standard deviation of Fp_{MLVSS}^{a} was 2.59 and Fp_{MLVSS}^{b} was 2.53., and 2-tailed sig. value obtained from *t-test* was 0.95 (>0.05). Mean difference between the data groups was 0.0583. These statistical analysis results clearly showed that measured and predicted P contents were very close to each other similar to the results reported in section 6.1.

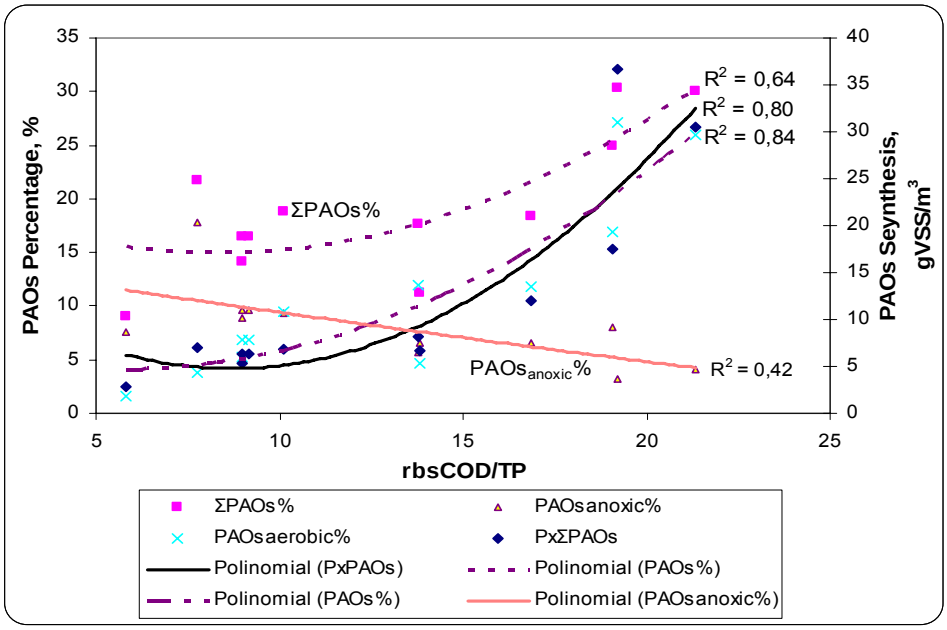


Figure 6.6 Effect rbsCOD/TP ratios on PAOs and activated sludge P content

EBPR efficiency was 30% at 10% PAOs domination, however gradually increase of PAOs percentage from 10% to 30%, EBPR efficiency reached to 88%. It is evident from Figure 6.6 that rbsCOD/TP ratio interacts with PAOs cell synthesis level and domination of PAOs within activated sludge system. It was found that the percentage of total PAOs (ΣPAOs) and aerobic PAOs, cell synthesis degree increased with increasing rbsCOD/TP ratios with determination coefficients of r^2=0.64; 0.80; 0.84. However anoxic PAOs percentage values showed a decreasing trend with increasing rbsCOD/TP ratio with a relatively smaller determination coefficient of r^2=0.42. This result may be explained with higher growth rates of

aerobic PAOs as compared to the anoxic PAOs. Higher energy yield of aerobic reactions as compared to anoxic reactions may also be another reason of population shift to aerobic PAOs.

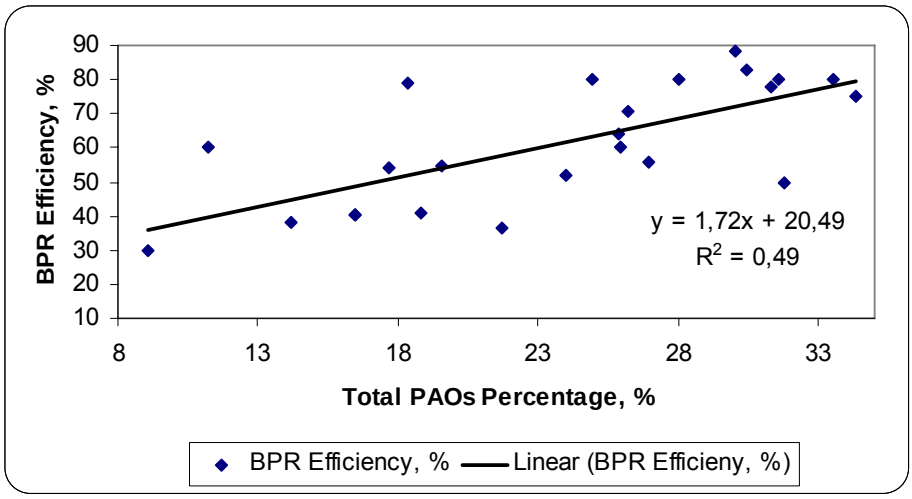


Figure 6.7 Relationship between EBPR efficiency and PAOs percentage

Evaluation of all full-scale investigation results given in Figure 6.7, indicated a linear interaction between EBPR efficiency and PAOs domination with a determination coefficient of 49%. This gradually lower correlation coefficient could be explained with interaction of EBPR performance with many parameters. Combined evaluation of Figure 6.3 and Figure 6.7 lead to an important conclusion that although sufficient level of PAOs exists in the activated sludge culture, their performance still depends on environmental and operational conditions including HRT_a, presence of electron acceptors in the anaerobic zone, sludge age etc. 30% percent PAOs domination would be sufficient to obtain approximately 90% EBPR efficiency unless system was limited by a parameter.

During the monitoring period mass fraction of PAOs with in the activated sludge ranged from 9% to 34% and mass fraction of non-PAOs microorganisms ranged from 81% to 66%. Moreover summer and winter mass fraction of PAOs was significantly different from each other. Average summer and winter intracellular phosphorus content was 16% and 25%. Investigation results also indicated that PAOs domination was proportional to rbsCOD/TP ratio and maximum domination was

detected at rbsCOD/TP ratio of 20. Microbial dispersal determined in this full scale study revealed that 30% PAOs domination resulted with an EBPR efficiency of 88%. Findings of this study also proposed that mass fraction of microbial groups in full scale systems could be completely different from laboratory scaled investigations. The reason of this difference could be explained with the fractions of COD used in bacterial cultivation. In the laboratory scaled investigations, influent contains only soluble COD forms. However influent of full scale systems contains significant amounts of particulate COD fractions. Most of these particulate COD forms could not be utilized by PAOs. Therefore preferential selection of these microorganisms may be limited by rbsCOD fraction of COD in full scale plants. Moreover mass fraction of OHOs could rise up to 90% within the activated sludge culture due to low influent soluble carbon forms.

During the monitoring period detailed batch scale activated sludge tests were also performed including anaerobic phosphorus release and VFAs uptake test, anoxic and aerobic phosphorus uptake tests. According to these batch tests, ratio of VFAs uptake rate to P release rate did not indicate significant seasonal variation. This ratio was 4.2 mgVFAs / mg P for summer period, 4.1 mgVFAs / mg P for winter period. Therefore it could be concluded that mass fraction of GAOs could not varied seasonally due to COD limited conditions in summer period. In pure cultures of PAOs, this ratio could be as low as 1.5 mgVFAs / mg P (Scruggs et. al., 2003). Therefore it could also be concluded that PAOs and GAOs could be nearly coexisted with in the activated sludge system.

CHAPTER SEVEN
IMPORTANCE of SUSPENDED BIODEGREDABLE MATERIALS in BIOLOGICAL PHOSPHORUS REMOVAL

7.1 Full – scale investigation results

In the last part of the thesis effect of biodegradable materials on EBPR was investigated. Two separate treatment lines, demonstrated in Figure 3.4 and 3.5, operating with and without PS (primary sedimentation) compared with each other mainly on a basis of EBPR and activated sludge characteristics. Influent and effluent of both two treatment lines were characterized by full scale measurements. Important process variables including sludge age, MLVSS, internal and external flow rates, SVI were monitored continuously. Obtained results were evaluated by both statistically and graphically. In addition to full-scale studies, activated sludge characteristics were investigated deeply using laboratory scaled batch tests described in the section 3.2.

According to the mechanism of EBPR, presence of VFAs in the anaerobic zone is vital to obtain a stable process performance. Since anaerobic contact period is too short for the conversion of particulate materials into soluble forms, it is believed that VFAs are derived from hydrolysis of simple substrate forms (USEPA, 1987; Metcalf & Eddy, 2003). However, statistical analysis results that were given in Table 7.1 and 7.2, indicated that effluent PO_4-P concentration of the treatment line operating without PS (Line III) was lower than the other treatment line operating with PS (Line I). Total 302 (151 data for Line I and 151 data for Line III) measurement results were statistically analyzed and it was found that the average PO_4-P concentration of the Line I was 3.8 mg/L and Line III was 1.6 mg/L. According to the Levene's test for equality of variances, sig. value for the PO_4-P concentrations of the two groups (Line I, Line III) was zero therefore equal variances was not assumed. Sig. (2-tailed) value obtained from the t-test for equality of means was also zero and 95% confidence interval of difference did not contain any zero. Therefore it could be safely concluded that the mean difference between PO_4-P concentrations of the two treatment lines were significant and effluent PO_4-P concentration of the Line III was

lower than the Line III averagely 2.12 mg/L. In addition to a lower effluent PO_4-P concentration, std. deviation of the Line III was 0.50 and Line I was 1.54 indicating a more stable EBPR performance.

Table 7.1 Group statistics of the experimental results recorded during the monitoring period

	Treatment Line	N	Mean	Std. Deviation	Std. Error Mean
Flow, m3/s	Line I	151	2,2130	,16691	,01358
	Line III	151	2,2184	,17417	,01417
MLVSS, mg/L	Line I	151	2,2130	240,144	19,543
	Line III	151	2650,71	581,012	47,282
Sludge Age, days	Line I	151	10,69	2,152	,175
	Line III	151	9,68	2,560	,208
SVI, ml/g	Line I	151	163,09	18,557	1,510
	Line III	151	123,61	34,557	2,812
Eff. PO4-P, mg/L	Line I	151	3,8221	1,54724	,12591
	Line III	151	1,6995	,50427	,04104

Good sludge settling characteristics of the activated sludge are essential for the entire effluent quality and SVI is one of the important indications of sludge settling behavior (Scruggs & Randall, 1998). Statistical analysis results showed that average SVI of the treatment lines were 163 ml/g for Line I and 123 ml/g for Line III. According to the Levene's test for equality of variances, sig. value for the SVI values of the two groups (Line I, Line III) was zero therefore equal variances was not assumed. Sig. (2-tailed) value obtained from the t-test for equality of means was also zero and 95% confidence interval of difference did not contain any zero. Therefore it could be safely concluded that the mean difference between SVI values were significant and SVI of Line III was lower than Line I, averagely 39 ml/g.

Both effluent PO_4-P concentrations and SVI values of the treatment lines were depicted in Figure 7.1 and Figure 7.2. Graphical evaluation of the investigation results clearly showed that operation without PS resulted with lower effluent PO_4-P concentrations and better sludge settling behavior. The reason(s) of these performance differences were investigated by scientific investigations.

Table 7.2 Statistical analysis of the experimental results recorded during the monitoring period

		Levene's Test for Equality of Variances		t-test for Equality of Means						
									95% Confidence Interval of the Difference	
		F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	Lower	Upper
Flow	Equal variances assumed	,100	,752	-,277	300	,782	-,00543	,01963	-,04406	,03320
	Equal variances not assumed			-,277	299,459	,782	-,00543	,01963	-,04406	,03320
MLVSS	Equal variances assumed	155,797	,000	-3,150	300	,002	-161,179	51,162	-261,860	-60,498
	Equal variances not assumed			-3,150	199,797	,002	-161,179	51,162	-262,065	-60,293
Sludge Age	Equal variances assumed	,323	,571	3,723	300	,000	1,013	,272	,478	1,549
	Equal variances not assumed			3,723	291,368	,000	1,013	,272	,478	1,549
SVI	Equal variances assumed	72,827	,000	12,367	300	,000	39,477	3,192	33,195	45,758
	Equal variances not assumed			12,367	229,865	,000	39,477	3,192	33,187	45,766
Effluent PO4-P	Equal variances assumed	106,625	,000	16,028	300	,000	2,12265	,13243	1,86204	2,38326
	Equal variances not assumed			16,028	181,511	,000	2,12265	,13243	1,86135	2,38395

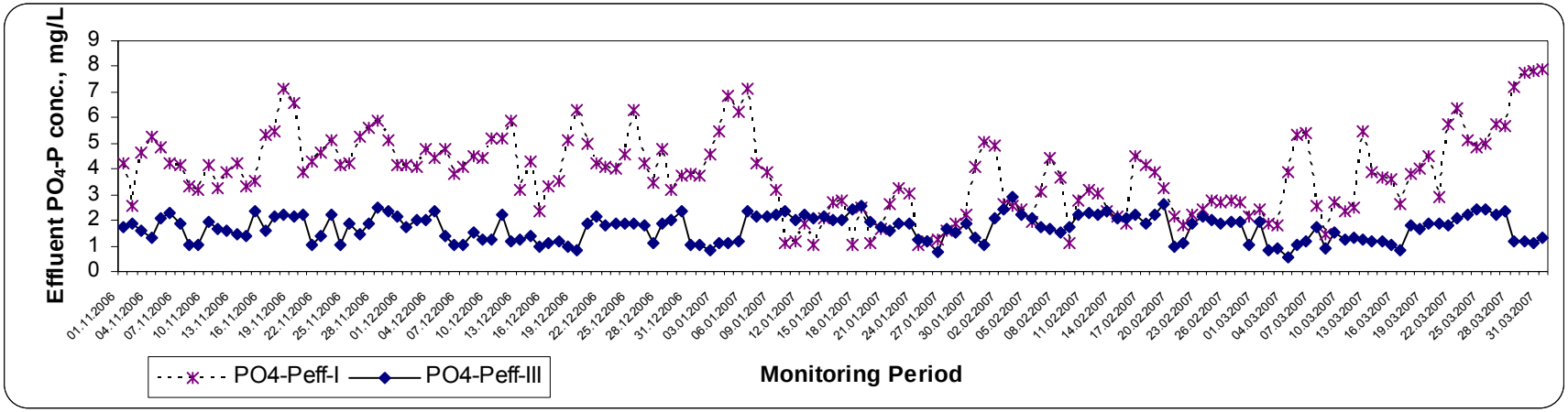


Figure 7.1 Effluent PO_4-P concentration of the treatment lines during the monitoring period

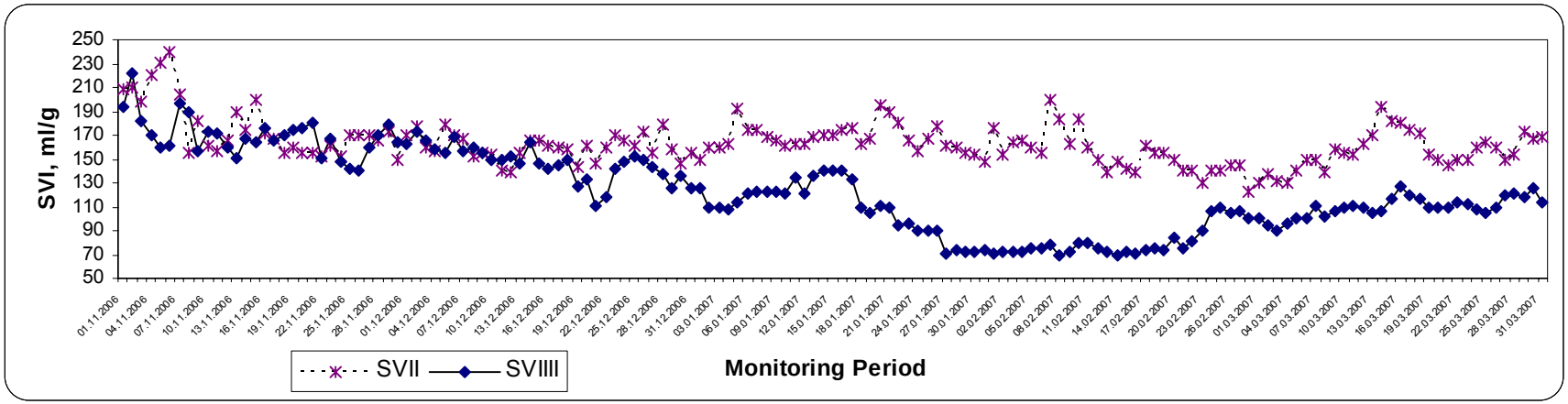


Figure 7.2 SVI of the treatment lines during the monitoring period

Previously studies showed that there are many parameters that could influence the EBPR process including wastewater composition (nutrient levels, pH, temperature), organic and hydraulic loading rates, presence of electron acceptors in the anaerobic zone, sludge age, MLVSS, internal and external flow rates. Therefore each of these factors have to be investigated scientifically using the background obtained from the previously studies.

Since the influent distributed to the lines (I-III) simultaneously there were no difference between the wastewater temperature of the treatment lines. During the monitoring period, wastewater temperature varied from 13.8 to 20.9 °C and with an average of 17.7 °C.

Table 7.3 Descriptive statistics of the influent wastewater temperature

	N (Data Number)	Minimum	Maximum	Mean	Std. Deviation
Wastewater Temperature, °C	151	13,80	20,90	17,75	1,420
Valid N (listwise)	151				

One of the important environmental factors is initial wastewater pH that is effective on both anaerobic and aerobic EBPR metabolism (Janssen et al., 2002; Liu et al., 1996; Converti et al, 1995; Liu Y., Chen Y., Zhou O., 2007). Initial pH of the treatment lines were continuously measured during the monitoring period and these data were statically analyzed to understand if there were significant difference.

Table 7.4 Group statistics of the initial pH values

	Grouping	N	Mean	Std. Deviation	Std. Error Mean
Influent pH	Line I	151	7,5507	,15752	,01282
	Line III	151	7,5519	,15621	,01271

According to the Levene's test results given Table 7.5, sig. value (0.853) for the initial pH values of the two groups (Line I, Line III) was higher than 0.05. Therefore equal variances were assumed. As it could be seen from Table 7.5, Sig. (2-tailed) value obtained from the t-test for equality of means was 0.947.

Table 7.5 Statistical analysis (Independent Samples T Test) result of the initial pH values

Equal variances assumed

	Levene's Test for Equality of Variances		t-test for Equality of Means						
								95% Confidence Interval of the Difference	
	F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	Lower	Upper
Initial pH	,035	,853	-,066	300	,947	-,00119	,01805	-,03672	,03434

Table 7.6 Statistical analysis (Independent Samples T Test) results of the initial COD, VFAs concentrations and rbsCOD/TP ratios

		Levene's Test for Equality of Variances		t-test for Equality of Means						
									95% Confidence Interval of the Difference	
	Assumptions	F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	Lower	Upper
COD	Equal variances assumed	11,120	,001	-10,148	226	,000	-190,969	18,819	-228,051	-153,886
	Equal variances not assumed			-11,919	221,720	,000	-190,969	16,023	-222,545	-159,393
rbsCOD	Equal variances assumed	,015	,904	-,225	152	,823	-1,442	6,415	-14,116	11,232
	Equal variances not assumed			-,225	151,991	,823	-1,442	6,415	-14,116	11,232
VFAs	Equal variances assumed	,041	,840	-,695	152	,488	-1,584	2,281	-6,090	2,922
	Equal variances not assumed			-,695	151,962	,488	-1,584	2,281	-6,091	2,922
rbsCOD/TP	Equal variances assumed	,267	,606	1,854	152	,066	2,0766	1,1199	-,1359	4,2891
	Equal variances not assumed			1,854	150,797	,066	2,0766	1,1199	-,1360	4,2893

It is evident from statistical analysis that the initial pH values of the treatment lines were not different from each other and it could also be concluded that PS did not affect the wastewater pH.

Organic and hydraulic loading rate to the biological treatment units are one of the important factors that could significantly affect the process performance. According to statistical test results comparing the flow rates of the two treatment lines indicated that (2-tailed sig.>0.05) difference between the data groups was not significant. As it could be seen from Table 7.2, the mean difference between the flow rates was 0.005 m^3/s. Graphical evaluation of the flow data of the treatment lines depicted in Figure 7.3-a) also indicated an overlap. Equivalence of the both internal and external recycle rates in addition to approximately equal flow rates could be concluded that hydraulic loading rates and so the hydraulic retention times of the treatment lines were very similar.

Although hydraulic loading rates were approximately equal, different organic loading rates could be expected since some part of the suspended organic materials removed in PS. Especially fractions of the COD could be influenced by PS due to hydrolysis and acidogenesis reactions. Previously studies showed that PS could be used to generate VFAs from settled organic materials with small revisions. However, in these systems, required SRT is higher than 1 day (Banister & Pretorius, 1996; Barajas et. al., 2001; Katehis et. al., 2003; McCue et. al., 2003). During the monitoring period, HRT in PS ranged from 1.4 to 2.2 h with an average of 2 h.

Table 7.7 Group statistics of the initial COD, VFAs concentrations and rbsCOD/TP ratios

	groupingII	N	Mean	Std. Deviation	Std. Error Mean
COD, mg/L	I Line	77	450,77	89,135	10,158
	III Line	151	641,74	152,265	12,391
rbsCOD, mg/L	I Line	77	193,86	39,959	4,554
	III Line	77	195,30	39,648	4,518
VFAs, mgHAC/L	I Line	77	75,19	14,264	1,625
	III Line	77	76,78	14,039	1,600
rbsCOD/TP	I Line	77	21,342	7,2523	,8265
	III Line	77	19,265	6,6310	,7557

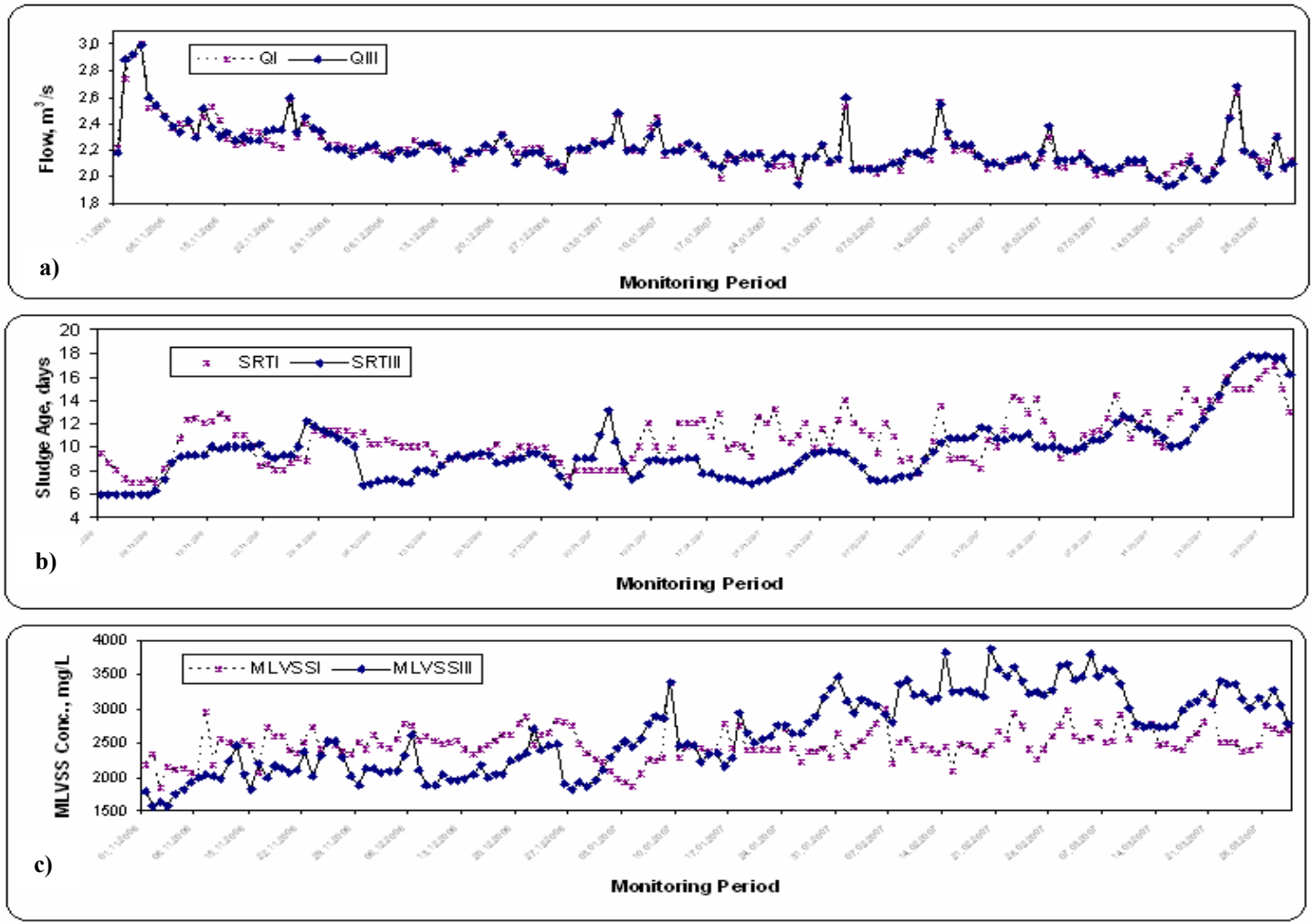


Figure 7.3 Flow rates, sludge ages and MLVSS concentrations of the treatment lines during the monitoring period

According to the statistical analysis of initial COD concentrations of the treatment lines given in Table 7.6, sig. value was 0.001 therefore equal variances were not assumed. Sig. (2-tailed) value obtained from the t-test for equality of means was also zero and 95% confidence interval of difference did not contain any zero. Therefore it could be concluded that the mean difference between initial COD concentrations were significant and initial COD concentration of Line III was higher than Line I averagely 190 mg/L due to PS. It could also be said that organic loading rates of the treatment lines were different from each other.

Sig. value obtained from Levene's test were higher than 0.05 (Table 7.6) so rbsCOD and VFAs concentration in the influent of Line I and Line III have the equal variances. Sig. (2-tailed) value obtained from the t-test for equality of means was 0.823 for rbsCOD and 0.488 for VFAs. These results could be concluded that soluble COD fractions (rbsCOD and VFAs) were not influenced by PS due to low HRT.

Since presence of electron acceptors in the anaerobic zone has a net reducing effect on soluble COD fractions, total concentrations of electron acceptors (influent: DO; effluent nitrate & DO) were defined as nitrate (a detailed description was given in section 3.1.6) for both two treatment lines to control if there were significant difference. According to the Table 7.8, average electron acceptor concentrations prior to the anaerobic zones (Line I & Line III) were 1.34 and 1.09 and their stand. deviations were 0.26 and 0.34. Although *t- test* results indicated a significant difference between the data groups (2-tailed sig.<0.05), mean difference between influent electron acceptor concentrations was 0.24 mg/L. Previous studies showed that 5 mg rbsCOD removed by 1 mg/L NO_3-N, therefore it could be concluded that influent electron acceptors level of the treatment lines could not influenced the effluent PO_4-P concentration significantly

Although rbsCOD concentrations in the influents of the treatment lines were very similar, rbsCOD/TP ratio of the treatment line operating with PS was slightly higher. Since some part of the particulate P (TP) is removed in PS, rbsCOD/TP ratio of the treatment line operating with PS was slightly higher than the other line.

Table 7.8 Group statistics of the influent electron acceptors (EA_{inf})

	Grouping	N	Mean	Std. Deviation	Std. Error Mean
EAinf	Line I	151	1,341	,2667	,0217
	Line III	151	1,095	,3785	,0308

Table 7.9 Group statistics of the influent TP concentrations

	Group	N	Mean	Std. Deviation	Std. Error Mean
TP	Line I	77	9,510	1,7338	,1976
	Line III	150	10,699	2,1512	,1756

Table 7.10 Statistical analysis (Independent Samples T Test) results of the influent electron acceptor concentrations (Equal variances was not assumed)

		Levene's Test for Equality of Variances		t-test for Equality of Means						
									95% Confidence Interval of the Difference	
		F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	Lower	Upper
EAinf	Equal variances assumed	4,759	,030	6,521	300	,000	,2457	,0377	,1716	,3198
	Equal variances not assumed			6,521	269,475	,000	,2457	,0377	,1715	,3199

Table 7.11 Statistical analysis (Independent Samples T Test) results of the influent TP concentrations

Equal variances assumed

	Levene's Test for Equality of Variances		t-test for Equality of Means						
								95% Confidence Interval of the Difference	
	F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	Lower	Upper
TP	2,309	,130	-4,199	225	,000	-1,1889	,2832	-1,7469	-,6310

Influent TP concentrations were also statistically analyzed and obtained results were given in Table 7.10 and Table 7.11. These results indicated that influents of the treatment lines were containing different concentration of TP (2-tailed sig.<0.05). Influent TP concentration of the Line III was higher than Line I averagely 1.18 mg/L. It is also evident from Table 7.6 and Table 7.7 that rbsCOD/TP ratio of the Line I was nearly 10% higher than the Line I. It could be concluded that the higher EBPR performance of the Line III was not due to soluble COD fractions.

Overall evaluation of the results lead to a conclusion that the organic loading rate difference were not causing from soluble COD fractions but causing from particulate biodegradable materials that had not been removed in the PS.

Sludge age is another critical parameter in optimization of the EBPR process as discussed previously Chapters. According to the group statistics given in Table 7.1, average sludge age of the Line I was 10.6 days and Line III was 9.6 days. Std. deviation of the Line I sludge age values was 2.152 and Line III was 2.560. Performed Levene's test (Table 7.2) also indicated that variances of the sludge age data could be assumed as equal (sig. value=0.571>0.05). 2-tailed sig. value obtained from t-test was zero indicating that the mean difference between sludge ages of the treatment lines was significant for a mean difference of 1 day. However this small difference could not affect the EBPR significantly. Lower and upper differences of the 95% confidence interval were 0.478 days (lower) and 1.549 days (upper) also indicated that variances of the sludge age values of the treatment lines could not create a significant EBPR performance difference. It is also evident from Figure 7.3 b. that both sludge ages of the treatment lines were always with in acceptable range during the monitoring period.

It was reported that in this type of EBPR process (5-Stage Modified Bardenpho), optimum MLVSS concentration should be with in the range of 2.5-3.0 g/L (USEPA, 1987; Metcalf & Eddy, 2003). MLVSS concentrations of the treatment lines during the monitoring period were also examined both statistically and graphically. According to the group statistics given in Table 7.1, average MLVSS concentration of Line I was 2.4 g/L and Line III was 2.6 g/L. Statistical analysis results were

presented in the Table 7.2. Since the sig. value that was obtained from Levene's test was zero therefore equal variances were not assumed and 2-tailed sig. value obtained from t-test analysis were 0.002. This result showed that the MLVSS concentrations of the treatment lines were different from each other and MLVSS concentration of the Line III was higher than Line I with an average of 0.161 g/L. Lower and upper differences of the 95% confidence interval were -0.262 g/L days (lower) and -0.060 g/L (upper). However this amount of MLVSS difference between treatment lines could not affect EBPR performance significantly. It is also evident from Figure 7.3 c. that both MLVSS concentrations of the treatment lines were good enough to obtain a stable EBPR process.

Both statistically and graphically evaluation of the full scale investigations showed that main difference in the treatments line's feeding and operational conditions that would influence the EBPR performance significantly was organic loading rate. Detailed wastewater characterization revealed that organic loading rate difference was originating from suspended materials that are not considered as an important EBPR performance criteria. Examination of the other important environmental and operational parameters was found relatively unimportant as compared to the organic loading rate. Therefore it could finally be concluded that not only soluble COD forms influence the EBPR performance but also particulate biodegradable materials also could play an important role in stabilization of the EBPR performance.

7.2 Batch – scale investigation results

In this section of the thesis, batch-scale experimental studies were conducted parallel to the full-scale investigations to both compare the characteristics of activated sludges obtained from the treatment lines (Line I, Line III) and to validate the results of full-scale investigations. Since previously studies clearly showed that the optimal carbon source was acetate, another carbon sources were not used in these tests (Brdjanovic, 1998; Pala & Bölükbaş, 2005). Batch tests were performed monthly during the last three months of the full-scale investigations. Anaerobic P release and sCOD utilization rates, anoxic P uptake and simultaneous denitrification

rate, aerobic P uptake rates of the two treatment lines were measured by these batch-scale tests.

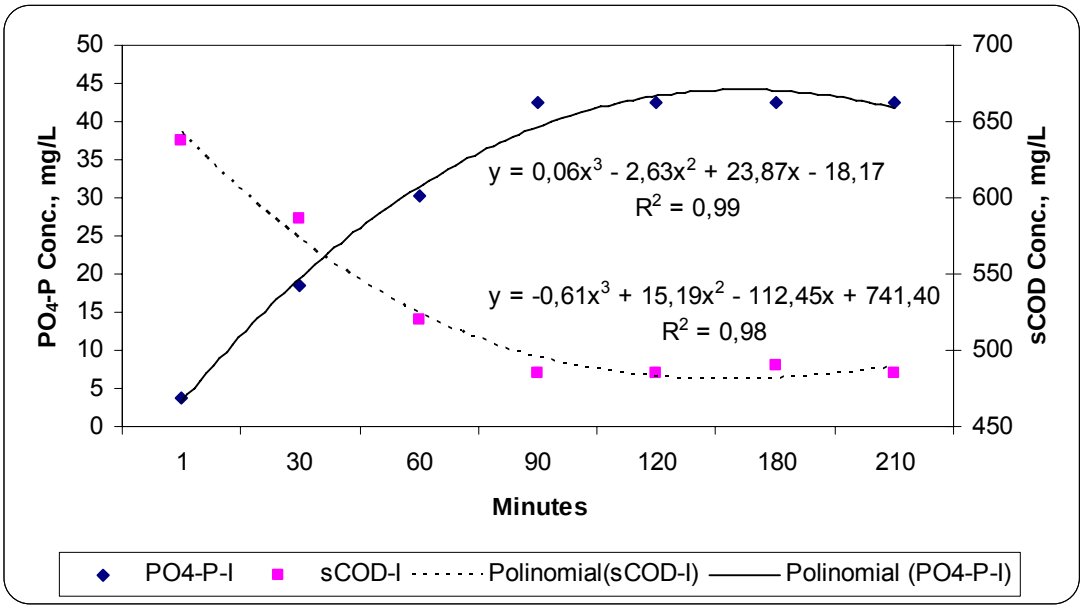


Figure 7.10 Anaerobic P release and sCOD utilization results (Line I – January)

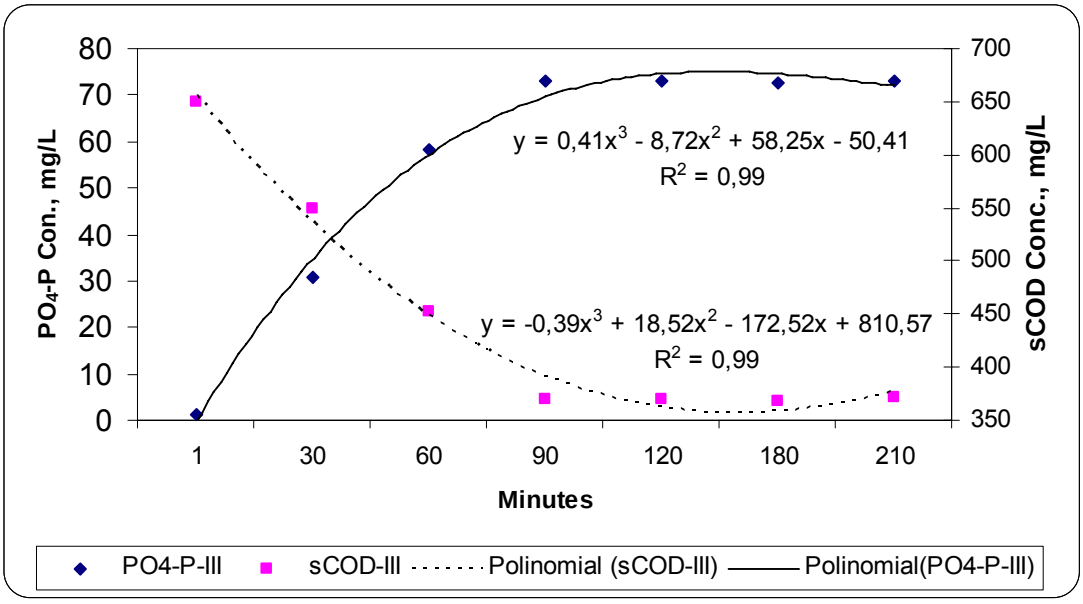


Figure 7.11 Anaerobic P release and sCOD utilization results (Line III – January)

Anaerobic P release and sCOD utilization profiles of the treatment lines were depicted in Figure 7.10 and 7.11. As it could be seen from above given figures, obtained results indicated a polynomial reaction kinetic with significant correlation

coefficients (r^2=0.99; 0.98 for Line I, r^2=0.99; 0.99 for Line III). P release rate of Line I was 0.19 mgP/gVSS/min and P release rate of Line III was 0.27 mgP/gVSS/min. P release rate of the Line III was higher than Line I approximately 70%. sCOD utilization rate of Line III was also higher than Line I approximately 71%. sCOD utilization rate of Line III was 0.76 mgsCOD/gVSS/min and Line I was 1.06 mgsCOD/gVSS/min.

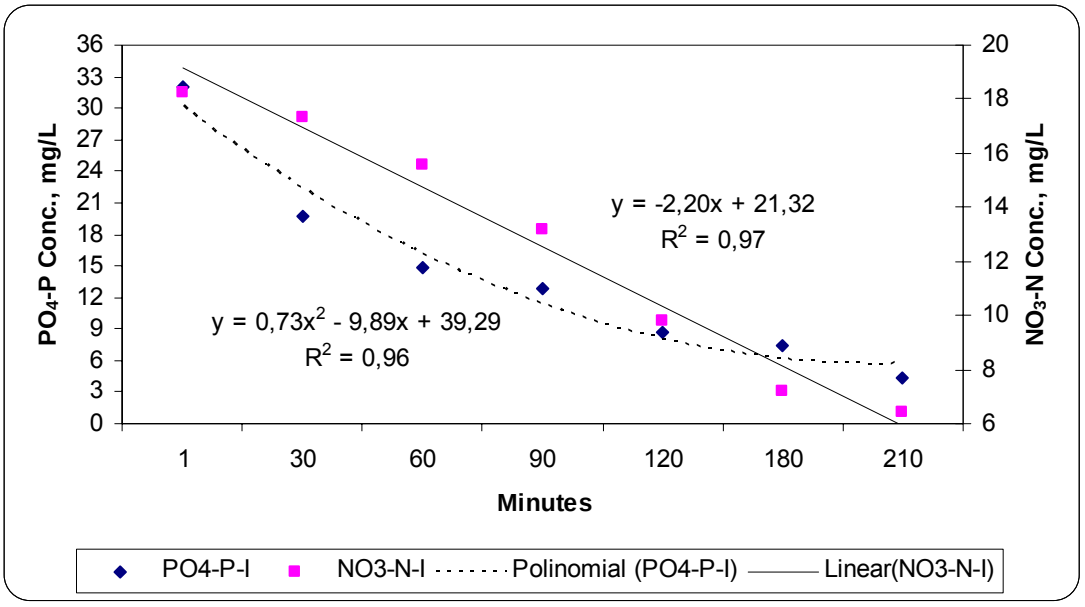


Figure 7.12 Anoxic P uptake and simultaneous denitrification results (Line I – January)

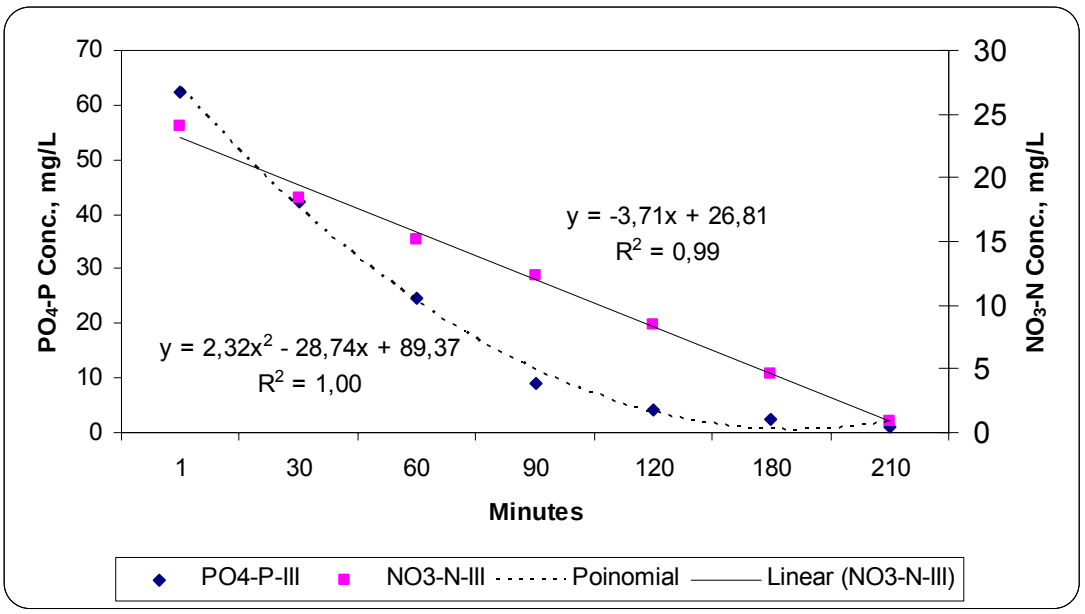


Figure 7.13 Anoxic P uptake and simultaneous denitrification results (Line III – January)

Anoxic P uptake and denitrification profiles of the treatment lines were depicted in Figure 7.12 and 7.13. As it could be seen from two figures (7.12 and 7.13), anoxic P uptake batch test results of both two treatment lines indicated a polynomial reaction kinetic with significant correlation coefficients (r^2=0.96 for Line I; r^2=1.00 for Line III). Anoxic P uptake rate of Line I was 0.06 mgP/gVSS/min and Line III was 0.10 mgP/gVSS/min. Therefore anoxic P uptake rate of the Line III was higher than Line I approximately 60%. It was interesting that nearly all of the released P (62 mg/L) was taken up by the activated sludge obtained from Line III during the anoxic conditions in less than 2.5 hours.

As it could be seen from two figures (7.12 and 7.13), denitrification reactions that happens simultaneously together with anoxic P uptake indicated a linear reaction kinetic with significant correlation coefficients (r^2=0.97 for Line I; r^2=0.99 for Line III). Denitrification rate of Line I was 0.03 mgNO_3-N/gVSS/min and Line III was 0.04 mgNO_3-N/gVSS/min. Therefore denitrification rate of the Line III was higher than Line I approximately 70%.

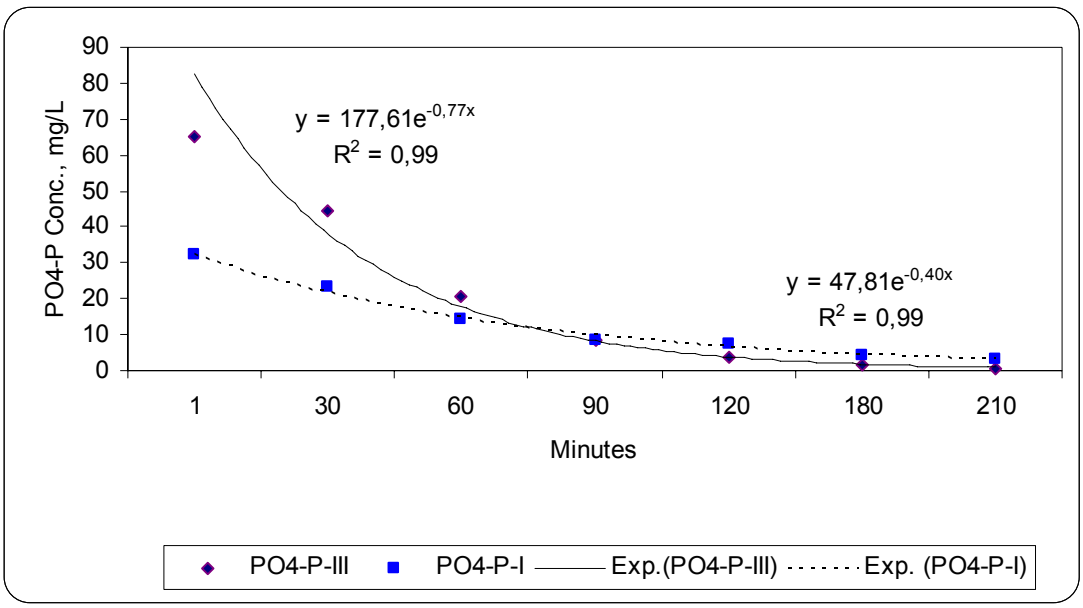


Figure 7.14 Aerobic P uptake results (Line I & III – January)

Figure 7.14 presents the aerobic P uptake results of both two lines. As it could be seen from the figure, aerobic P uptake measurement results of the two activated sludge sample showed an exponential reaction kinetics with a correlation coefficient of r^2=0.99. Aerobic P uptake rate of Line I was 0.06 mgP/gVSS/min and Line III was 0.10 mgP/gVSS/min. The aerobic P uptake rate of Line III was higher than Lin I approximately 60% exactly the same with anoxic P uptake.

All batch-scale experimental results including anaerobic P release and sCOD utilization, anoxic P uptake and denitrification, aerobic P uptake rates were in accordance with each other and strongly supporting the theory of that all these reactions (anaerobic/anoxic/aerobic) could be carried out by the same microorganisms. These results were also validating the full-scale experimental results.

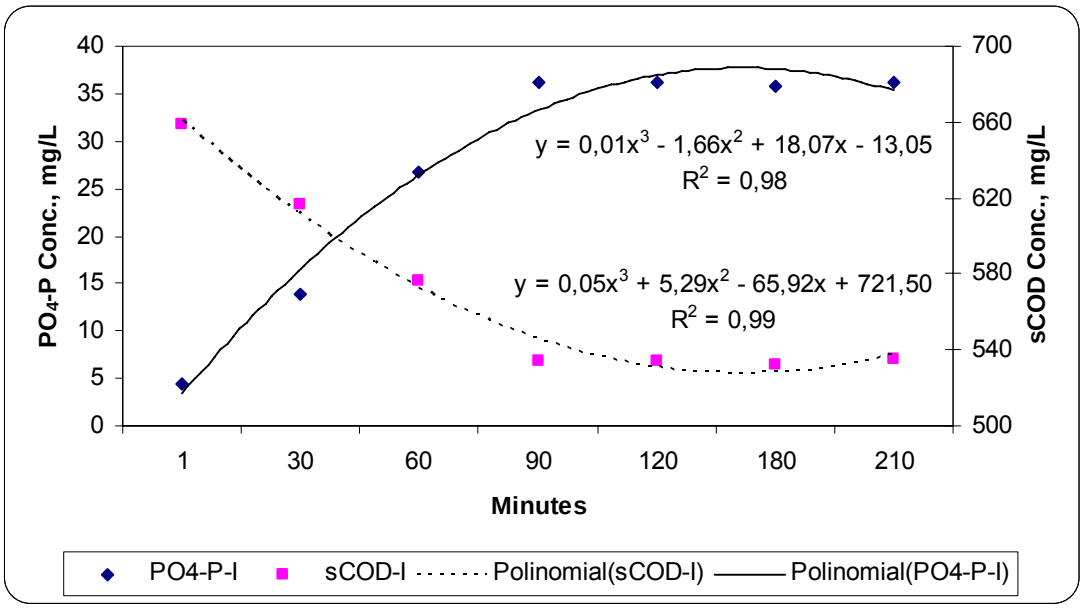


Figure 7.15 Anaerobic P release and sCOD utilization results (Line I – February)

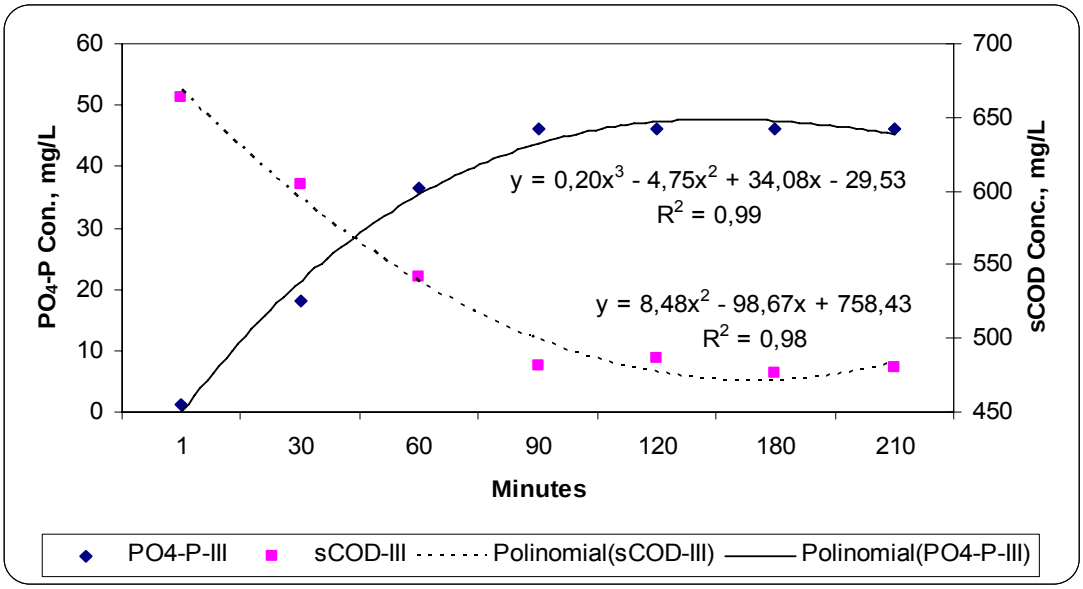


Figure 7.16 Anaerobic P release and sCOD utilization results (Line III – February)

Anaerobic P release and sCOD utilization profiles of the treatment lines were depicted in Figure 7.15 and 7.16. As it could be seen from the figures, obtained results indicated a polynomial reaction kinetic with significant correlation coefficients (r^2=0.98; 0.99 for Line I, r^2=0.99; 0.98 for Line III). P release rate of Line I was 0.20 mgP/gVSS/min and P release rate of Line III was 0.27 mgP/gVSS/min. P release rate of the Line III was higher than Line I approximately 74%. As expected, sCOD utilization rate of Line III was also higher than Line I approximately 70%. sCOD utilization rate of Line III was 0.78 mgsCOD/gVSS/min and Line I was 1.11 mgsCOD/gVSS/min.

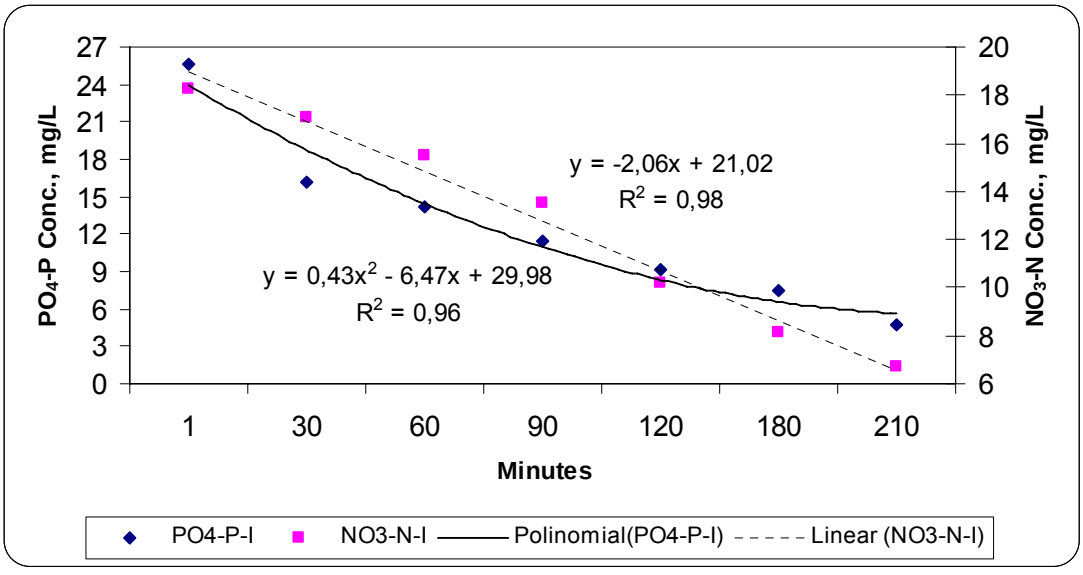


Figure 7.17 Anoxic P uptake and simultaneous denitrification results (Line I – February)

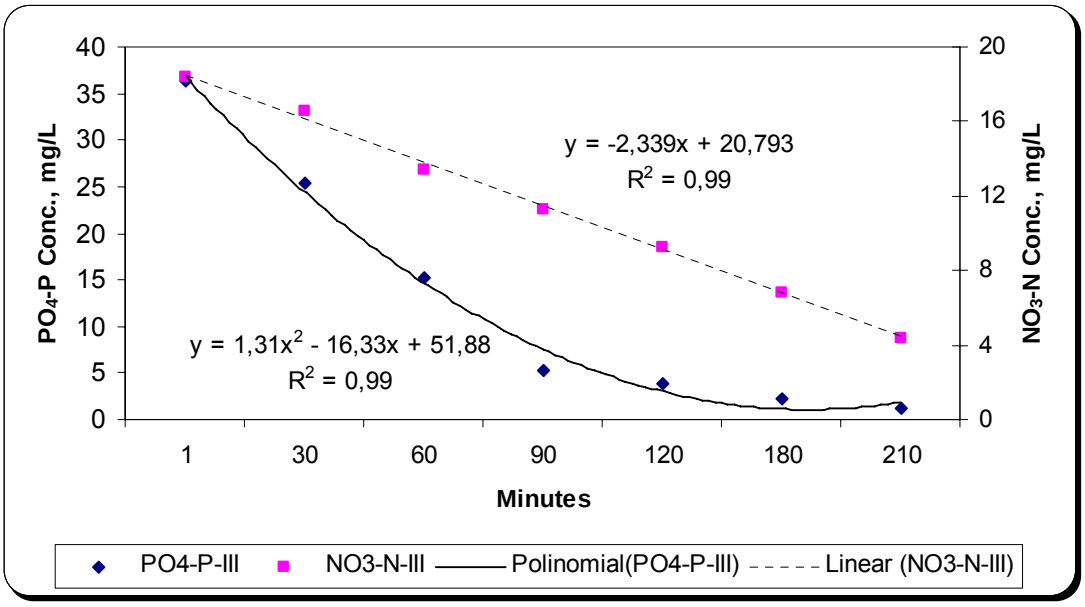


Figure 7.18 Anoxic P uptake and simultaneous denitrification results (Line III – February)

Anoxic P uptake and denitrification profiles of the treatment lines were depicted in Figure 7.17 and 7.18. Similar to the results obtained from previous experimental set (January) anoxic P uptake batch test results of both two treatment lines indicated a polynomial reaction kinetic with significant correlation coefficients (r^2=0.96 for Line I; r^2=0.98 for Line III). Anoxic P uptake rate of Line I was 0.06 mgP/gVSS/min and Line III was 0.10 mgP/gVSS/min. Therefore anoxic P uptake rate of the Line III was higher than Line I approximately 60%. Nearly all of the released P (36 mg/L) was taken up by the activated sludge obtained from Line III during the anoxic conditions in less than 2.5 hours similar with the previous experiments.

As it could be seen from two figures (7.17 and 7.18), denitrification reactions that happen simultaneously together with anoxic P uptake indicated a linear reaction kinetic with significant correlation coefficients (r^2=0.98 for Line I; r^2=0.99 for Line III). Denitrification rate of Line I was 0.03 mgNO_3-N/gVSS/min and Line III was 0.04 mgNO_3-N/gVSS/min. Therefore denitrification rate of the Line III was higher than Line I approximately 70%.

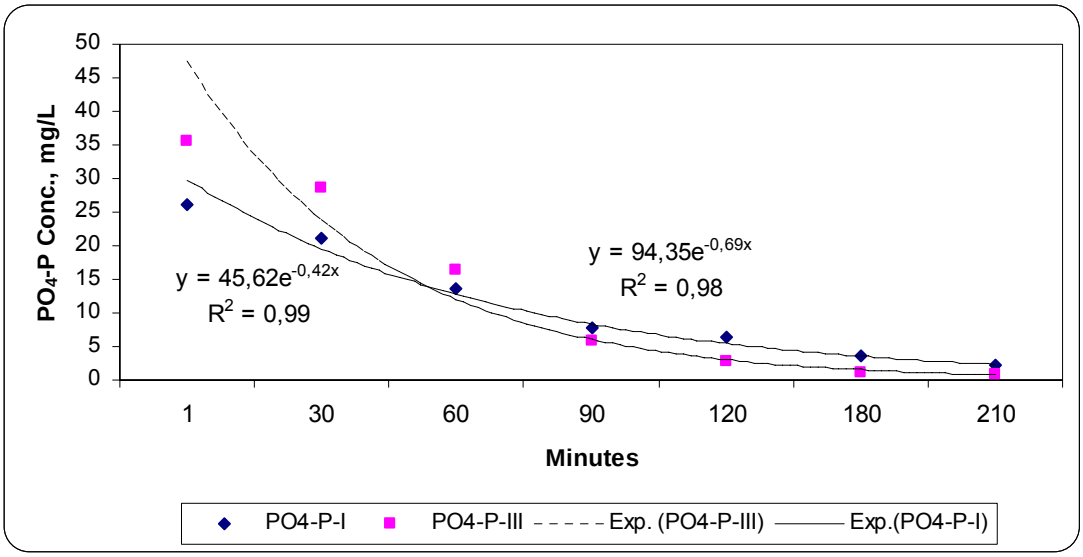


Figure 7.19 Aerobic P uptake results (Line I & III – February)

Figure 7.19 presents the aerobic P uptake results of both two lines. As it could be seen from the figure, aerobic P uptake measurement results of the two activated sludge samples showed an exponential reaction kinetics with an correlation coefficients of r^2=0.99 for Line I and r^2=0.98 for Line III. P uptake rate of Line I was 0.07 mgP/gVSS/min and Line III was 0.10 mgP/gVSS/min. The aerobic P uptake rate of Line III was higher than Lin I approximately 70% similar to P uptake rate in anoxic period.

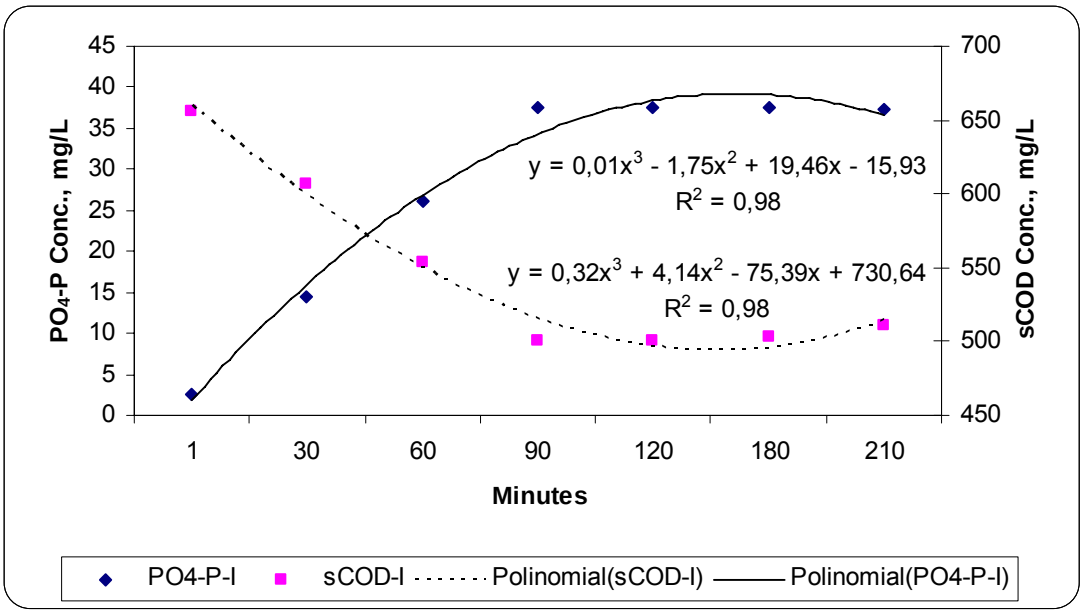


Figure 7.20 Anaerobic P release and sCOD utilization results (Line I – March)

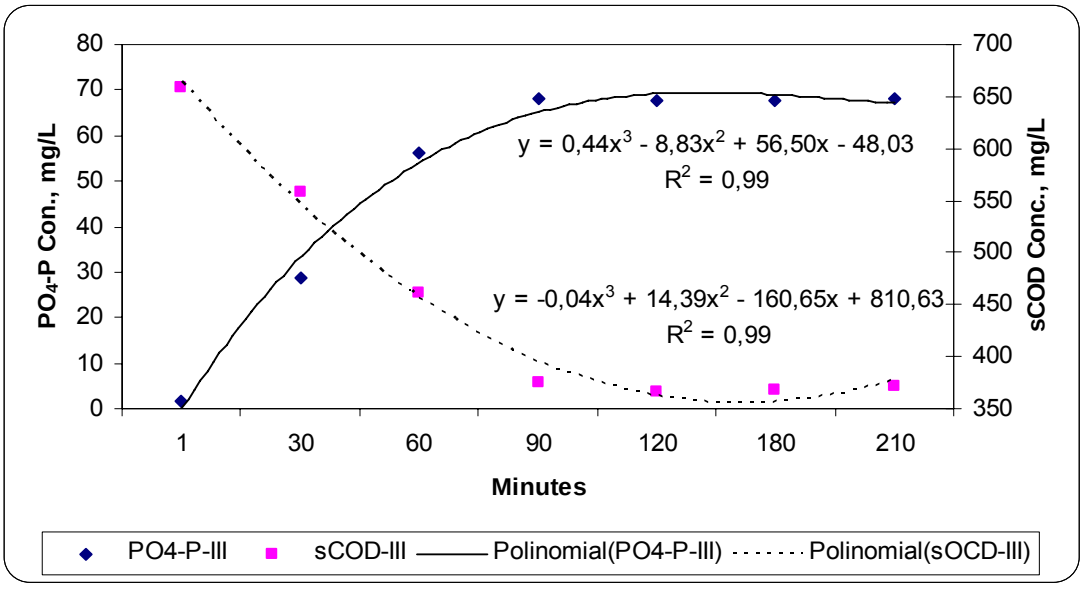


Figure 7.21 Anaerobic P release and sCOD utilization results (Line III – March)

Anaerobic P release and sCOD utilization profiles of the treatment lines were depicted in Figure 7.20 and 7.21. Obtained results indicated a polynomial reaction kinetic with significant correlation coefficients (r^2=0.98; 0.98 for Line I, r^2=0.99; 0.99 for Line III). P release rate of Line I was 0.14 mgP/gVSS/min and P release rate of Line III was 0.27 mgP/gVSS/min. P release rate of the Line III was higher than Line I approximately 51%. As expected, sCOD utilization rate of Line III was also higher than Line I approximately 54%. sCOD utilization rate of Line I was 0.62 mgsCOD/gVSS/min and Line III was 1.13 mgsCOD/gVSS/min.

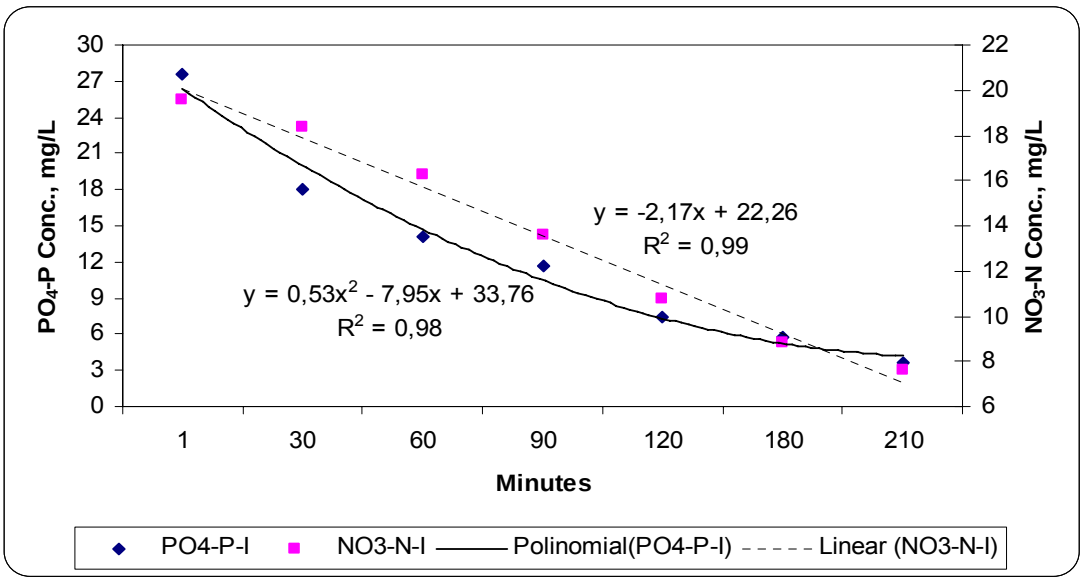


Figure 7.22 Anoxic P uptake and simultaneous denitrification results (Line I – March)

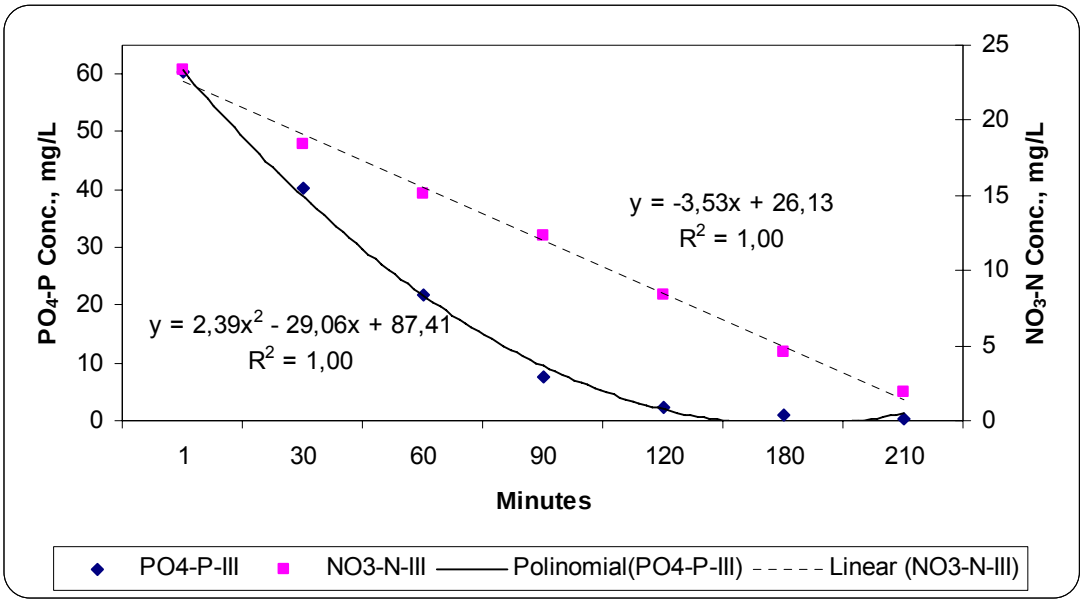


Figure 7.23 Anoxic P uptake and simultaneous denitrification results (Line III – March)

Anoxic P uptake and denitrification profiles of the treatment lines were depicted in Figure 7.22 and 7.23. As it could be seen from two figures, anoxic P uptake batch test results of both two treatment lines indicated a polynomial reaction kinetic with significant correlation coefficients (r^2=0.98 for Line I; r^2=1.00 for Line III). Anoxic P uptake rate of Line I was 0.05 mgP/gVSS/min and Line III was 0.10 mgP/gVSS/min. Therefore anoxic P uptake rate of the Line III was higher than Line I approximately 50%. Similar to the previously experimental results (January & March) nearly all of the released P (60 mg/L) was taken up by the activated sludge obtained from Line III during the anoxic conditions in less than 2.5 hours.

Denitrification reactions that happens simultaneously together with anoxic P uptake that were depicted in Figure 7.22 and Figure 7.23, indicated a linear reaction kinetic with significant correlation coefficients (r^2=0.99 for Line I; r^2=1.00 for Line III). Denitrification rate of Line I was 0.03 mgNO_3-N/gVSS/min and Line III was 0.04 mgNO_3-N/gVSS/min. Therefore denitrification rate of the Line III was higher than Line I approximately 70%.

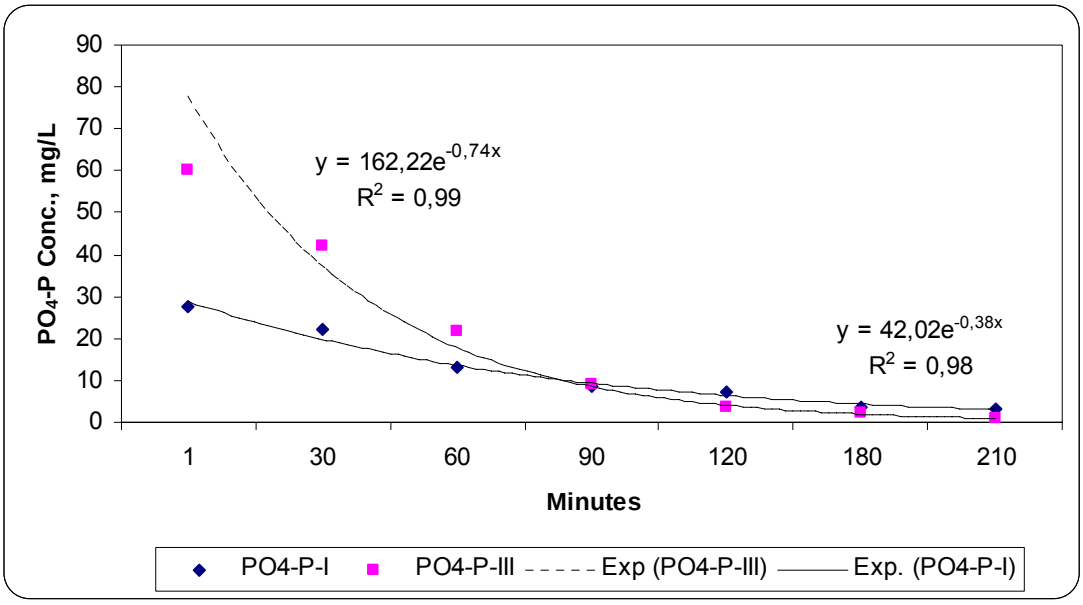


Figure 7.24 Aerobic P uptake results (Line I & III – March)

Figure 7.24 presents the aerobic P uptake results of both two lines. As it could be seen from the figure, aerobic P uptake measurement results of the two activated sludge sample showed an exponential reaction kinetics with an correlation coefficients of r^2=0.99 for Line I and r^2=0.98 for Line III. P uptake rate of Line I was 0.05 mgP/gVSS/min and Line III was 0.10 mgP/gVSS/min. The aerobic P uptake rate of Line III was higher than Lin I approximately 50% similar to P uptake rate in anoxic period.

Table 7.12 Summary of the batch-scale investigation results

Parameter	Average		Unit	January 2007		February 2007		March 2007	
	Line I	Line III		Line I	Line III	Line I	Line III	Line I	Line III
Anaerobic P Release Rate	*0,18*	*0,27*	mgP/gVSS/min	0,19	0,27	0,20	0,27	0,14	0,27
Anaerobic sCOD Utilization Rate	*0,72*	*1,10*	mgsCOD/gVSS/min	0,76	1,06	0,78	1,11	0,62	1,13
sCOD consumed / P released	*4,00*	*4,08*	mgsCOD/mgP	3,94	3,86	3,88	4,09	4,18	4,30
Anoxic P Uptake Rate	*0,06*	*0,10*	mgP/gVSS/min	0,06	0,10	0,06	0,10	0,05	0,10
Denitrification Rate	*0,03*	*0,04*	mgNO_3-N/gVSS/min	0,03	0,04	0,03	0,04	0,03	0,04
NO_3-N consumed / P uptake	*0,49*	*0,37*	mgNO_3-N/mgP	0,42	0,37	0,55	0,39	0,50	0,35
Aerobic P Uptake Rate	*0,06*	*0,10*	mgP/gVSS/min	0,06	0,10	0,07	0,10	0,05	0,10

EBPR characteristics of the activated sludge that were determined using batch experiments were summarized in the Table 7.12. Similar to the full-scale experimental results, batch-scale experimental results clearly showed that EBPR reaction kinetics of the Line III was much more faster than Line I. Anaerobic P release rate, ability of the activated sludge to utilize COD in anaerobic zone, anoxic P and aerobic uptake rates and denitrification rate of the Line III were higher than Line I more than 60% with an average. It is evident from these faster reaction rates that the EBPR performance difference between the treatment lines were not only causing from metabolic P removal as a result of particulate COD degradation but also were causing from stabilizing effect of biodegradable suspended materials on both effluent PO_4-P concentration and SVI. An idealized behavior of the activated sludge under anaerobic and aerobic conditions was also shown in Figure 7.25.

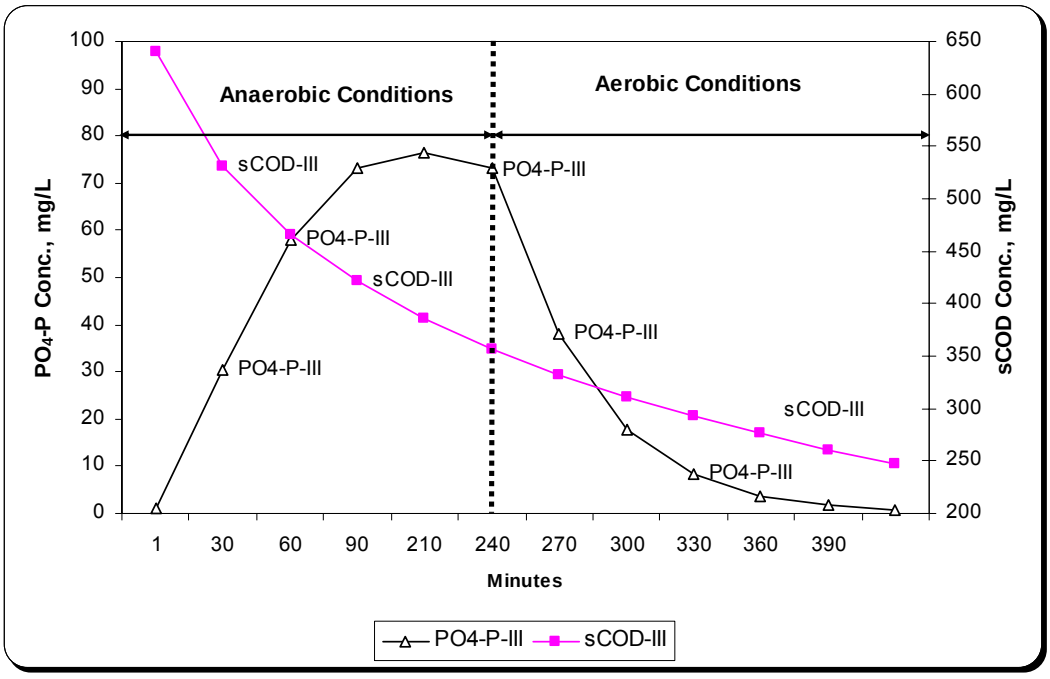


Figure 7.25 Idealized PO_4-P and sCOD profile of the activated sludge (Line III)

As it could be seen from Table 7.12, experimental studies conducted on the two activated sludge samples (Line I & III) indicated that 4.0 mg sCOD is removed by 1 mg P released. However nitrate requirements of the activated sludge samples to remove 1 mg P were different from each other. While Line I was using 0.49 mg Nitrate, Line III was using 0.37 mg Nitrate to remove 1 mg P from the bulk interface averagely. This result also indicated an improved reaction yield of 75% averagely. The higher yield efficiency could be explained with excellent domination of PAOs.

CHAPTER EIGHT
MANAGEMENT STRATEGIES FOR IMPROVMENT of PHOSPHORUS REMOVAL in WASTEWATER TREATMENT PLANTS

8.1 Determination of phosphorus removal method

In this section of the thesis, obtained results from experimental studies and reviewed literature was combined to determine the most reliable method to control the effluent phosphorus level. As discussed deeply in Chapter 2, phosphorus could be removed by both biological and chemical methods. In a conventional biological treatment system 10-30% of phosphorus present in the influent could be removed. However addition of an anaerobic selector before aerated zones could enhance phosphorus removal efficiency up to 90% (USEPA, 1987; Metcalf & Eddy, 2003). Removal of phosphorus by these enhanced EBPR systems became one of the standard wastewater treatment applications. Many scientific researches indicated that EBPR has many advantages as compared to the chemical phosphorus removal (CPR) methods.

CPR systems could create an increase in total sludge production because of improved suspended and dissolved solids removal efficiency, formation of metal – hydroxide precipitants etc. As a result of increased daily sludge amount, cost of sludge treatment and disposal increases as well. (Janssen, Meinema, van der Roest, 2002). In addition to increased sludge volume, dewaterability characteristics of the sludge could be deteriorated as a result of chemical addition. In EBPR systems, phosphate is preserved as potassium or magnesium phosphate (polyphosphate) without adhered water, resulting in a good dewaterable product (Janssen, Meinema, van der Roest, 2002).

Use of metal salts and lime for phosphorus removal not only negatively affects sludge dewatering characteristics but also influence other sludge disposal unit operations including anaerobic sludge digestibility, thermal drying and incineration of the sludge (USEPA, 1987). Since chemically removed sludge contains more

adhered water, moisture content is higher than the non-chemical sludge. Therefore in combination of increased sludge volume and higher sludge moisture content, energy requirement that is one of the most determinative factors in operational costs will be significantly higher as compared to non-chemical sludges from the point of sludge mixing, pumping, and heating (anaerobic digestion, thermal drying and incineration).

It was also reported that CPR is increased salinity and conductivity level in effluent originating from remaining negative ions (chloride and sulfide). In addition to increased salinity, effluent total dissolved solids (TDS) concentration could be increased as well due to impurities present in the metal salts (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

In many full-scale WWTPs phosphorus and nitrogen is removed simultaneously. Addition of metal salts could destroy the wastewater buffer capacity that could result with inhibited or poorer nitrification efficiency especially for wastewater with weak buffer capacity. Dosing of chemicals will remove the available COD not only for PAOs but also for the denitrifiers that means lowered denitrification efficiency. It could be concluded that CPR processes could deteriorate overall nitrogen removal efficiency (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

Phosphorus is one of the non-renewable and limited natural sources. Moreover, increased industrial and agricultural phosphorus demand increased the market price of the phosphorus (Isherwood, 2000). Investigations showed that recovery of chemically precipitated phosphorus is not possible or at least not feasible (Donnert and Salecker, 1999a; Donnert and Salecker, 1999b). In contrary, recovery of phosphorus from wastewater with an efficiency range of (10% -80%) is only possible with EBPR processes in which sludge became rich in phosphate (Strickland, 1999; Gaterell et al., 2000; Jeanmaire and Evans, 2001). It was also reported that fertilizer effect of EBPR sludge is attractive resource for agricultural purpose (Meinema, van der Roest, 2002).

In addition to these advantages of biological methods for removal of phosphorus from wastewaters, many scientific investigations were also demonstrated that EBPR processes are highly dependent upon influent wastewater characteristics such as; COD/TP ratio, pH and temperature etc. Moreover sludge handling systems should be carefully designed and well operated to prevent phosphorus releases back to the plant (Meinema, van der Roest, 2002; Metcalf & Eddy, 2003). Therefore use of properly designed and operated enhanced EBPR methods should be the primary alternative and use of chemical methods should be considered as secondary alternative to always guarantee effluent phosphorus level below legal discharge limits in WWTPs.

8.2 Optimization of EBPR in WWTPs

8.2.1 Establishing mass balances

To evaluate performance of existing wastewater treatment facilities accurately, establishing mass balances all-around treatment facilities is important not only for phosphorus removal but also for all pollution parameters. In this section of the thesis, mass balances based on loading rate were presented schematically to trace the phosphorus pathways in the WWTP. In the following figures a representative overview of the phosphorus recirculation was demonstrated.

As it could be seen from the Figure 8.1, 82 tons of phosphorus (both in particulate and soluble forms) enters to the anaerobic tanks. As result of anaerobic reactions, serious amount of phosphorus is released back to the bulk interface and total phosphate amount increases from 4.6 tons/d to 31 tons/d. Therefore 26.4 tons of phosphorus releases in the form of orthophosphate in daily basis. Phosphorus fraction in the MLVSS also decreases from 3.3% to 2.1%.

In the following anoxic and aerobic zones 27.7 tons of orthophosphate is taken up by the bacteria and phosphorus fraction in the MLVSS increases from 2.1% to 3.5% as result of this phosphorus luxury uptake. After this oxidative environment, 82 tons of total phosphorus (with fractions is loaded to the final clarifiers as demonstrated in Figure 8.2.

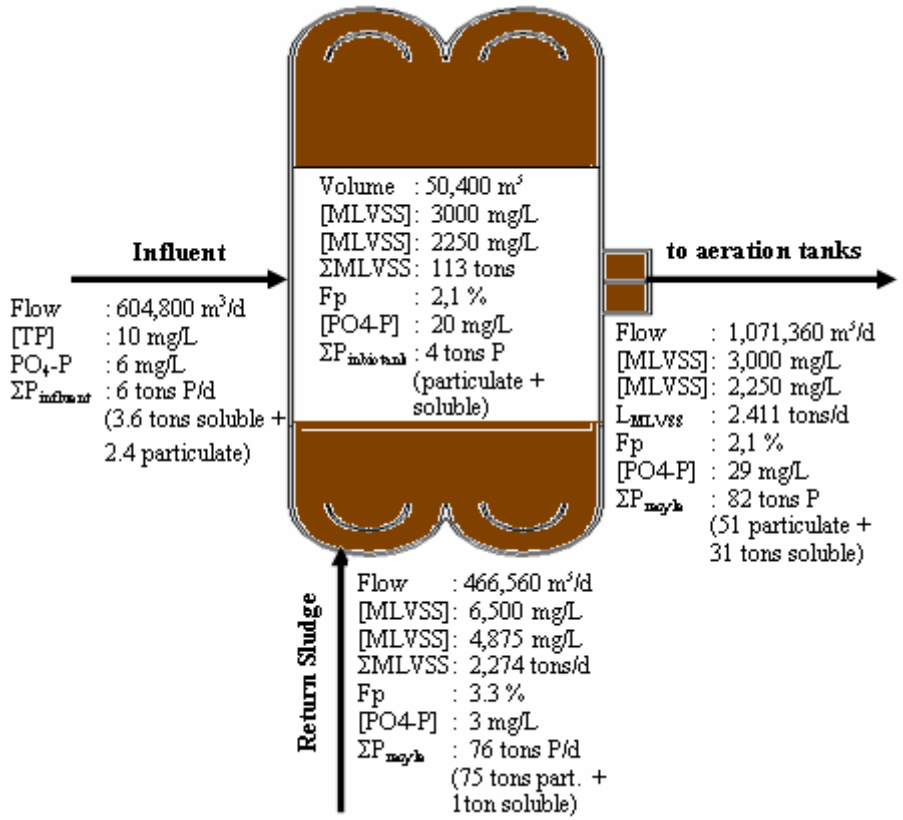


Figure 8.1 Mass balance around anaerobic tanks

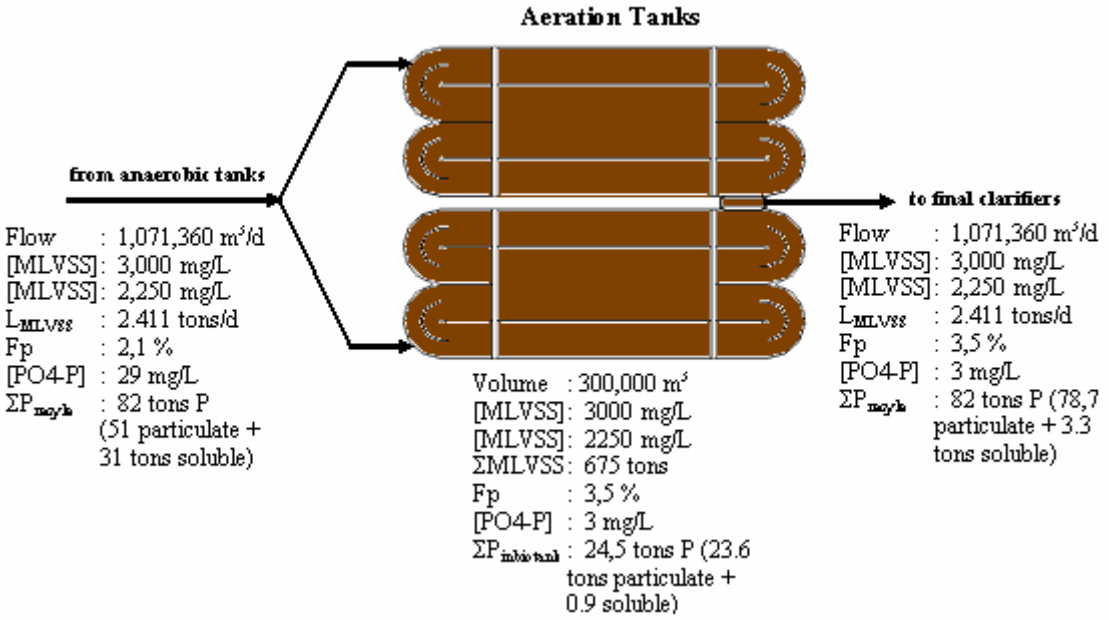


Figure 8.2 Mass balance around aeration tanks

Total 82 tons of phosphorus including 78.7 particulate and 3.3 tons soluble forms is loaded to the final clarifiers. As it could be seen from the Figure 8.3, 2.5 tons of phosphorus goes to effluent, 76 tons is recycled to the anaerobic tank via return sludge line and the remaining 3.5 ton of phosphorus is transferred to the excess sludge equalization tank.

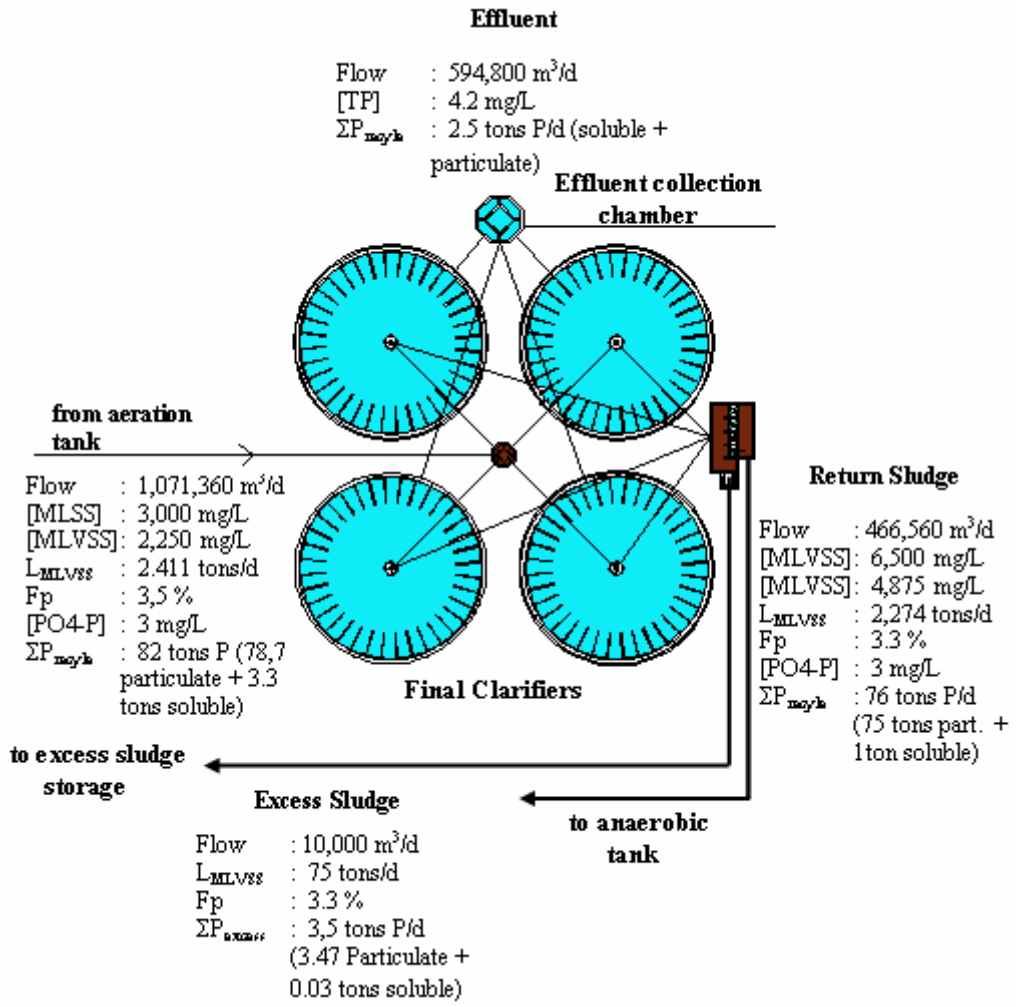


Figure 8.3 Mass balance around final clarifiers, return and excess sludge pumping stations

According to the wastewater characteristics of the WWTP, averagely 3,000 tons of primary sludges (PS) produced per day with a average dry solids (DS%) content of 2%. However DS content of the sludge could vary significantly according to the wastewater characteristics and sludge withdrawal frequency. Averagely 0.6 tons of phosphorus is removed in PS daily and transferred to the PS storage tank as demonstrated in Figure 8.4.

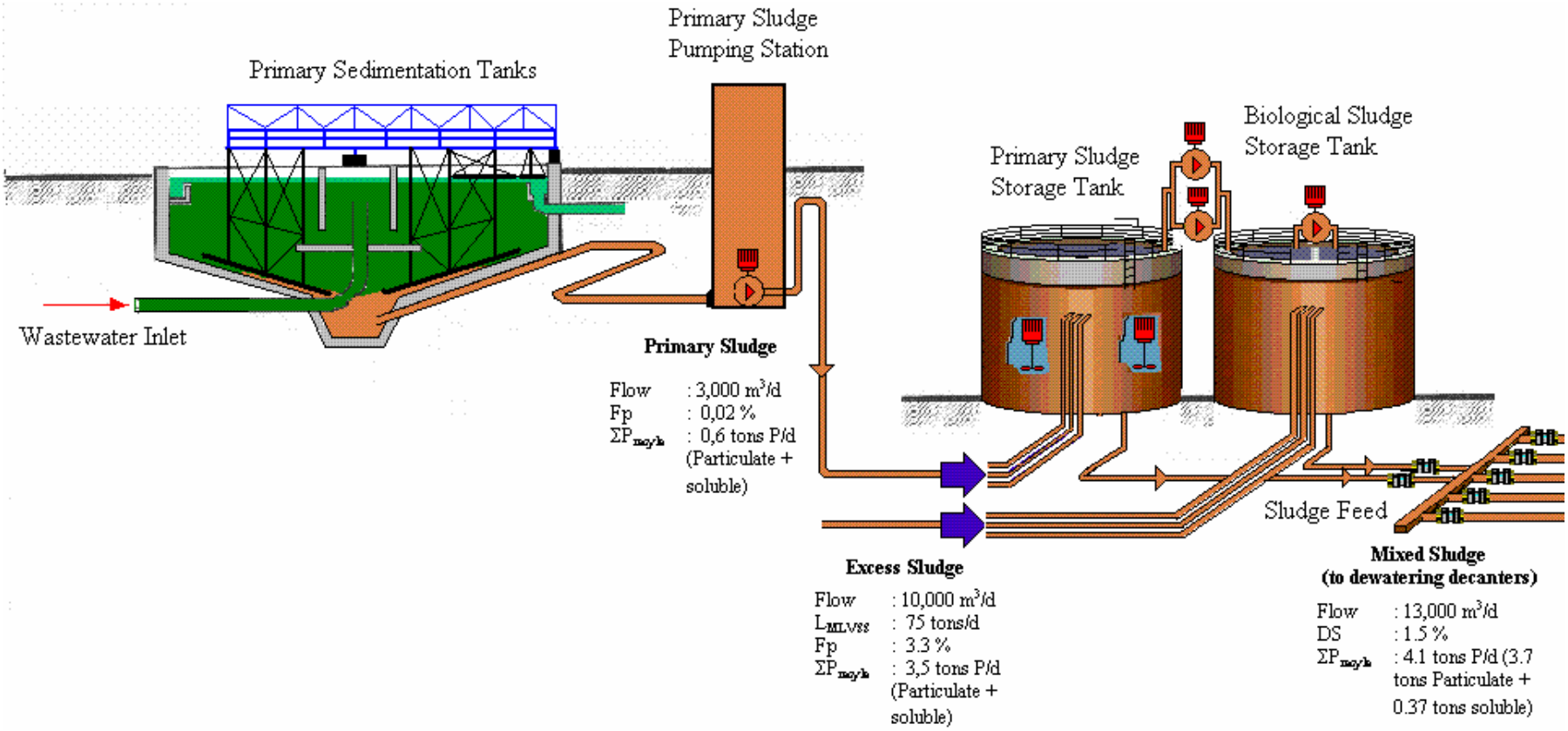


Figure 8.4 Mass balances around primary sedimentation and sludge equalization tanks

Primary and excess sludge were mixed in the excess sludge storage tank to improve the dewatering efficiency. Average DS content of this mixed sludge is 1.5%. After mixing, sludge is sucked to the sludge thickening – dewatering system which consists of mainly 7 decanters. As it was demonstrated in the Figure 8.4, 4.1 tons of phosphorus including 3.7 tons particulate and 0.37 tons soluble fraction is fed to the decanter system. This mass balance approach indicated that significant amount of phosphorus is released back in the sludge storage tank in which primary and excess sludge is mixed. According to the mass balance results, nearly 0.34 tons of phosphorus released from the sludge to the liquid interface. The reason of this undesired phosphate increase is insufficient aeration in the tank. Dissolved oxygen concentration in the tank is averagely 0.5 mg/L. This result indicated that adequate aeration system should be equipped in sludge storage or sludge handling systems of the WWTPs with EBPR process configuration. Long term operational experiences indicated that aeration of these sludge holding tanks should be performed by jet – aerator mixers providing both air and mixing from the point of regular maintenance flexibility. In fact, membrane diffusers are one important alternative; however their required maintenance works is more difficult as compared to the jet mixers. In these mixer-aerators, required air could be supplied from the impeller without additional compressor or blower devices.

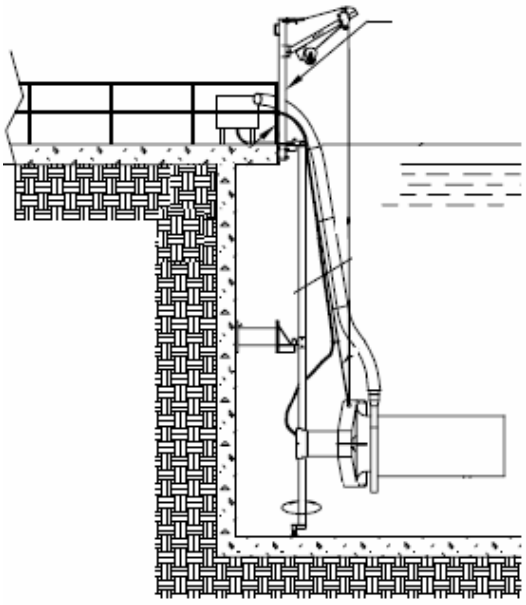


Figure 8.5 Cross - sectional view of a typical aerator - mixer

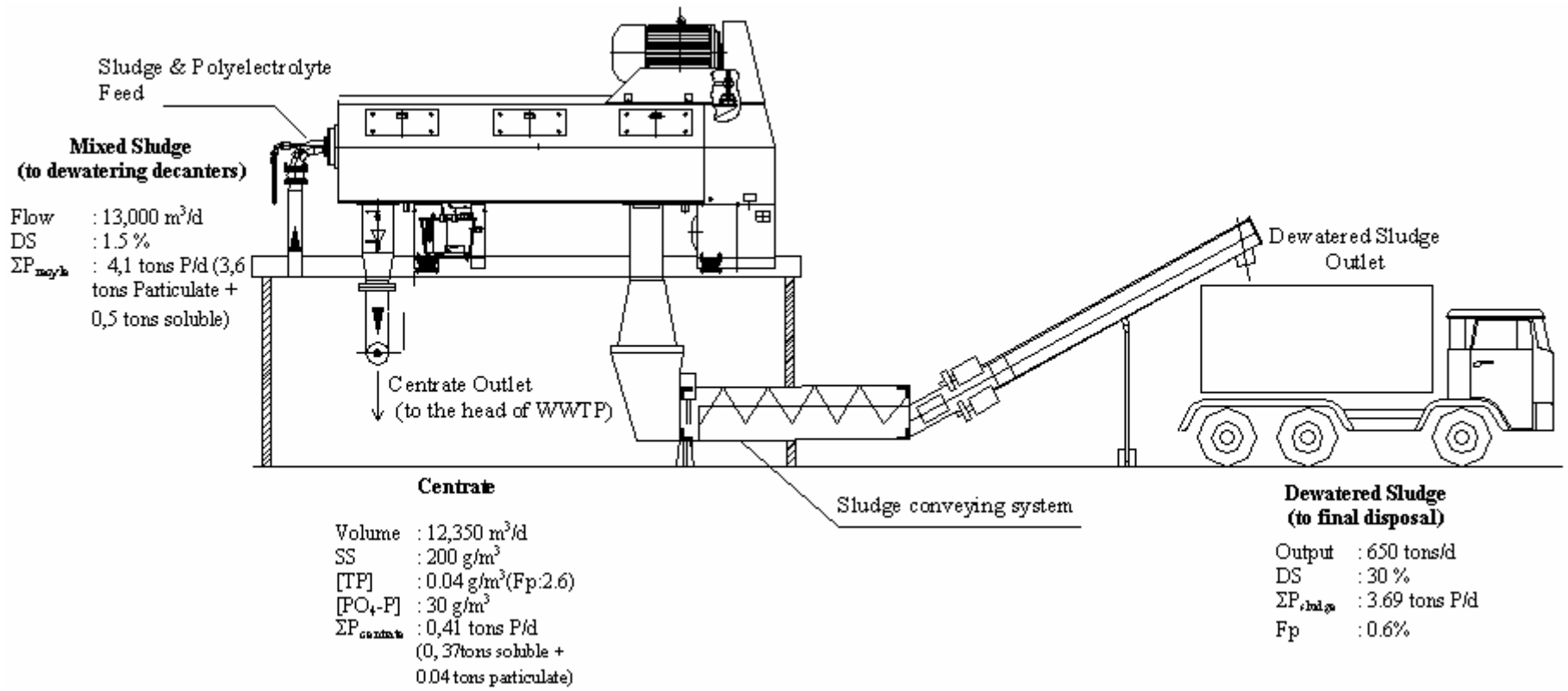


Figure 8.6 Mass balances around sludge dewatering units

In the current sludge handling system of the WWTP, raw sludge is directly dewatered without thickening and sludge DS content increases from 1.5% to 30%. As it was demonstrated in the Figure 8.6, 13,000 m^3/d raw sludge is dewatered and as a result of this application, 650 tons of sludge with a DS content of 30% and 12,350 m^3 centrate phases come out. Phosphorus fraction of this dewatered sludge is 0.6% and contains 3.69 tons of total phosphorus. As it could be seen from the Figure 8.6, 0.41 tons of phosphorus was lost with in the sludge handling facilities and send back to head of the WWTP.

Although there has not been introduced any legal phosphorus discharge limitation yet, if target effluent phosphorus concentration is assumed as 2 mg/L that means 1.2 tons P/d discharge is allowed, additional phosphorus removal requirement is 1.3 tonsP/d. (It was depicted in the Figure 8.3 that, mass balance results indicated 2.5 tons P/d effluent load). If phosphorus reload from sludge handling facilities (0.41 tons/d) is compared with required phosphorus removal degree (1.3 tons/d), it could be clearly seen that nearly 30% of improvement in phosphorus removal efficiency is possible with only preventing phosphorus reloads from side streams originating from sludge handling systems.

There are several ways to eliminate this side stream phosphorus loads, including aeration of sludge holding tanks and chemical precipitation of the phosphorus existing in the centrate (or filtrate) using several metal salts. However aeration alternative have several advantages as compared to chemical methods including elimination of odor in the sludge holding tanks and elimination of negative effects of chemical sludges on sludge handling facilities.

In the above given sections, a general overview of the management strategies in WWTPs were discussed on general layout using average mass balance results. In the following sections, whole experimental results obtained from the full scale investigations were expressed as mass balances that were based on loading rate.

Table 8.1 Mass balances performed around anaerobic tanks

Flow	PO_4-P_{inf}	PO_4-P_{rxl}	PO_4-$P_{anaerobiceff}$	LPO_4-P_{inf}	LPO_4-P_{rxl}	ΣLPO_4-P_{inf}	LPO_4-$P_{anaerobiceff.}$	Net P Release
m^3/d	mg/L	mg/L	mg/L	kg/d	kg/d	kg/d	kg/d	kg/d
155.520	5,4	2,0	10,5	840	311	1.151	3.266	2.115
190.080	5,8	2,7	11,6	1.102	420	1.522	4.009	2.487
146.880	4,5	0,6	11,1	661	93	754	3.357	2.602
241.920	5,4	3,5	9,2	1.306	544	1.851	3.656	1.806
155.520	4,2	0,6	11,6	653	93	746	3.608	2.862
155.520	5,0	0,6	11,2	778	93	871	3.484	2.613
241.920	6,2	3,9	11,0	1.500	607	2.106	4.372	2.265
198.720	5,1	1,7	10,5	1.013	264	1.278	3.720	2.442
155.520	3,9	0,7	7,7	607	103	709	2.395	1.686
241.920	5,5	3,8	9,5	1.331	591	1.922	3.776	1.854
190.080	9,1	3,9	20,0	1.730	607	2.336	6.912	4.576
198.720	5,4	2,1	10,6	1.073	327	1.400	3.755	2.355
181.440	4,4	1,0	11,6	798	156	954	3.909	2.955
138.240	6,1	4,5	9,1	843	700	1.543	2.673	1.130
146.880	5,1	4,1	7,6	749	638	1.387	2.298	912
138.240	4,9	4,6	8,1	677	715	1.393	2.379	987
155.520	6,2	5,4	7,4	937	840	1.777	2.270	492
155.520	4,8	4,2	7,1	746	653	1.400	2.208	809
155.520	4,8	4,2	7,1	746	653	1.400	2.208	809
155.520	4,3	2,2	7,5	669	342	1.011	2.333	1.322
155.520	5,6	4,1	9,2	871	638	1.509	2.862	1.353
155.520	6,4	2,1	13,6	968	327	1.294	4.171	2.877
155.520	7,5	1,3	19,9	1.134	202	1.336	6.104	4.768
155.520	9,1	2,1	24,0	1.415	327	1.742	7.465	5.723

LPO_4-P_{inf}: *Phosphate load from influent;* ***LPO_4-P_{rxl}:*** *Phosphate load from rxl;* ***ΣLPO_4-P_{inf}:*** *Total phosphate load to anaerobic tanks;* ***LPO_4-$P_{anaerobiceff.}$:*** *Total phosphate output from anaerobic tanks*

In the Table 8.1, mass balances established around anaerobic tanks were given as loading rates. According to the measurement results, averagely 1,391 kg of phosphate enters to the anaerobic tanks from both influent and return sludge, and the phosphate output from the tanks increases to 3.633 kg/d. This result indicates that 2,242 kg of phosphorus is released through the anaerobic reactions daily.

Table 8.2 Mass balances performed around anoxic zone

$\Sigma LPO_4\text{-}P_{input}$	**$PO_4\text{-}P_{int}$**	**$LPO_4\text{-}P_{int}$**	**$\Sigma LPO_4\text{-}P_{anoxiczone}$**	**$PO_4\text{-}P_{anoxic}$**	**$LPO_4\text{-}P_{anoxiceff}$**	**Anoxic P Uptake**
kg/d	**mg/L**	**kg/d**	**kg/d**	**mg/L**	**kg/d**	**ton/d**
3.266	2,8	658	3.924	6,5	3.549	375
4.009	3,5	823	4.831	6,5	3.774	1.058
3.357	0,4	94	3.451	3,7	1.988	1.462
3.656	3,6	846	4.502	6,2	3.921	581
3.608	0,8	188	3.796	4,5	2.457	1.339
3.484	1,2	282	3.766	4,0	2.184	1.581
4.372	5,0	1.175	5.547	6,6	4.174	1.373
3.720	1,8	423	4.143	4,0	2.357	1.786
2.395	0,6	141	2.536	3,3	1.802	734
3.776	4,4	1.034	4.810	5,9	3.731	1.078
6.912	3,6	846	7.758	12,1	7.025	733
3.755	2,3	541	4.295	4,9	2.887	1.408
3.909	0,9	212	4.120	5,5	3.146	974
2.673	4,7	1.105	3.778	5,4	2.855	922
2.298	4,0	940	3.238	5,1	2.741	497
2.379	4,7	1.105	3.484	5,7	3.014	470
2.270	5,2	1.222	3.492	5,8	3.142	350
2.208	4,1	964	3.172	5,1	2.785	387
2.208	4,1	964	3.172	5,1	2.785	387
2.333	2,4	564	2.897	4,5	2.457	440
2.862	4,0	940	3.802	5,9	3.222	580
4.171	2,1	494	4.665	6,9	3.738	927
6.104	1,0	235	6.339	10,4	5.634	705
7.465	2,1	494	7.958	13,4	7.317	641

$\Sigma LPO_4\text{-}P_{input}$: Total phosphate load from influent; $LPO_4\text{-}P_{int}$: Phosphate load from internal recycle; $LPO_4\text{-}P_{anoxiceff}$: Phosphate output from anoxic zone

According to the measurement results, averagely 3,633 kg of phosphate enters to the anoxic zone from influent and 679 kg from internal recycle. Therefore total average phosphate loading rate to the anoxic zone is 4,311 kg/d. As a result of anoxic reactions, averagely 866 kg of phosphate is uptake by bacteria and stored in

intracellular environment. Remaining 3,445 kg of phosphate in the liquid phase enters to the aerobic zone.

Table 8.3 Mass balances performed around aerobic zone

LPO_4-$P_{anoxiczone}$	PO_4-$P_{aerobic}$	LPO_4-$P_{aerobiceff}$	Aerobic P Uptake
kg/d	mg/L	mg/L	ton/d
3,549	2,8	871	2,678
3,774	3,5	1,210	2,564
1,988	0,4	121	1,867
3,921	3,6	1,431	2,490
2,457	0,8	249	2,208
2,184	1,2	373	1,811
4,174	5,0	1,987	2,187
2,357	1,8	638	1,719
1,802	0,6	187	1,615
3,731	4,4	1,749	1,983
7,025	3,6	1,244	5,781
2,887	2,3	815	2,073
3,146	0,9	303	2,843
2,855	4,7	1,381	1,475
2,741	4,0	1,210	1,531
3,014	4,7	1,381	1,633
3,142	5,2	1,595	1,547
2,785	4,1	1,275	1,510
2,785	4,1	1,275	1,510
2,457	2,4	746	1,711
3,222	4,0	1,244	1,978
3,738	2,1	644	3,094
5,634	1,0	307	5.27
7,317	2,1	653	6,664

As it could be seen from the Table 8.3, 2,492 kg of phosphate removed in the aerobic zone by the bacteria. Estimations given in the Table 8.2 indicated that, total phosphate loading rate from the anoxic zone to aerobic zone is 3,445 kg/d, therefore 954 kg of phosphate remains in the mixed sludge liquor and fed to the final clarifiers.

Table 8.4 Mass balances performed aeration tanks (anoxic and aerobic zone)

Date	ΣLPO_4-P	Anoxic P Uptake	Aerobic P Uptake	LPO_4-$P_{effluent}$
	kg/d	kg/d	kg/d	kg/d
10 Jan 06	3,924	375	2,678	871
20 Jan 06	4,831	1,058	2,564	1,210
14 Feb 06	3,451	1,462	1,867	121
22 Feb 06	4,502	581	2,490	1,431
28 Feb 06	3,796	1,339	2,208	249
02 Mar 06	3,766	1,581	1,811	373
14 Mar 06	5,547	1,373	2,187	1,987
02 Apr 06	4,143	1,786	1,719	638
21Apr 06	2,536	734	1,615	187
01 May 06	4,810	1,078	1,983	1,749
22 May 06	7,758	733	5,781	1244
03 Jun 06	4,295	1,408	2,073	815
28 Jun 06	4,120	974	2,843	303
10 Jul 06	3,778	922	1,475	1,381
26 Jul 06	3,238	497	1,531	1,210
02 Aug 06	3,484	470	1,633	1,381
31 Aug 06	3,492	350	1,547	1,595
08 Sep 06	3,172	387	1,510	1,275
27 Sep 06	3,172	387	1,510	1,275
14 Oct 06	2,897	440	1,711	746
08 Nov 06	3,802	580	1,978	1,244
15 Nov 06	4,665	927	3,094	644
24 Nov 06	6,339	705	5,327	307
30 Nov 06	7,958	641	6,664	653

ΣLPO_4-P: Total phosphate load to the aeration basin, LPO_4-$P_{effluent}$: Total phosphate output from the biological treatment units.

In the Table 8.4, a general overview of the aerobic EBPR reactions was given. These mass balances were established according to the measurement results and it is evident from the Table 8.4 that in each one of the experimental sets, balances were closed with zero difference. Form another point of view; following equation were always confirmed.

$$\Sigma L_{PO_4-P} = AnocixPUptake + AerobicPUptake + LPO_4 - P_{effluent}$$

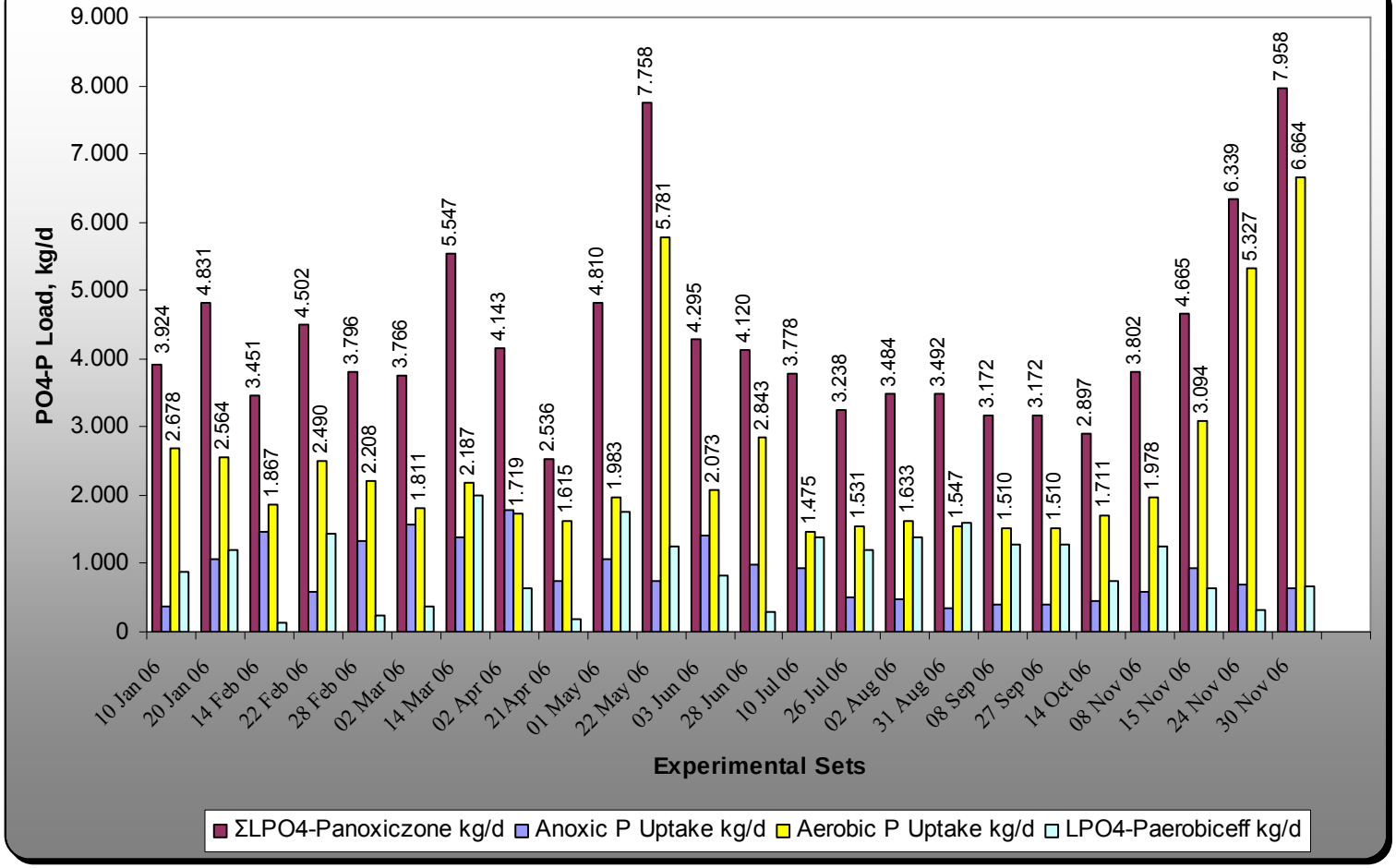


Figure 8.7 Balanced phosphate amounts around aeration tanks

8.2.2 Optimization of COD/TP ratio in WWTPs

As discussed previously in Chapter 2, one of the most critical concerns in maintaining a stabile EBPR is ratio between COD and phosphorus. Moreover, carbonaceous constituents present in wastewater could play an important role in EBPR. Laboratory scaled batch experiments clearly indicated that 4 mg VFAs is required to remove 1 mg of phosphorus. As it was proved by above given mass balance results that additional phosphorus requirement is 1.3 tons/d. Furthermore, it was also estimated that approximately 30% percent phosphorus removal could be achieved by preventing secondary releases from sludge handling system; therefore nearly 1 tons of phosphorus should be removed by biological methods daily.

8.2.2.1 Elimination of electron acceptor input to the anaerobic zone

According to the measurement results, minimum VFAs concentration (to simulate worst condition) in the influent is nearly 47 mg HAC/l that correspond 29 ton HAC/d loading rate. In addition, 2 tons/d of electron acceptor (as nitrate equivalent) inputs to the anaerobic tanks and as result of that, 10 ton/d VFAs is utilized in the denitrification reactions (5 mg VFAs/NO_3-N). Therefore 19 ton/d VFAs is available for the PAOs that sufficient to remove 4.7 ton P/d. As it could be seen from the Figure 8.1, total amount of phosphorus entering to the EBPR is 6 ton/d. Therefore, additional phosphorus removal requirement is estimated as 1.3 ton/d. This result also confirms the estimated requirement from mass balances.

As it was discussed in the above given section, nearly 34% percent of simple substrate forms (VFAs) available in the influent that is vital for proliferation of PAOs, consumed in undesired denitrification reactions. From another point of view, elimination of electron acceptor input to the EBPR system with a %50 efficiency will be sufficient to approach target effluent phosphorus level. Therefore elimination of electron acceptor input to the anaerobic zone is very critical concern for full scale WWTPs. Especially, this concern should be taken into consideration at design level.

Full scale investigations indicated that dissolved oxygen is one of the important electron acceptors and mainly carried to the anaerobic tank by both influent and return sludge. It could be safely concluded that high velocity gradients and turbulence caused oxygenation of both influent and return sludge streams. Therefore hydraulic design of return sludge transfer system and wastewater distribution chamber existing on the head of anaerobic tank should be well-designed to eliminate turbulences in both side and main streams.

Nitrate is another significant electron acceptor and measurement results indicated that concentration of nitrate in the return sludge line could increase up to 4 mg/L. To eliminate nitrate entrance to the anaerobic tank, post denitrification zone should be well designed and should have sufficient volume to provide enough contact time for nitrate utilization in which carbon source are derived from endogenous respiration.

8.2.2.2 VFAs production from prefermentation of primary sludge

As it was discussed in section 2.4, prefermentation could be used to generate VFAs to improve EBPR efficiency in WWTPs with weak wastewater characteristics. Many investigation results showed that production of VFAs by prefermentation could be a cost effective method (USEPA, 1987; Janssen, Meinema, van der Roest, 2002).

According to the revised literature sources a process configuration given in Figure 8.4 is sufficient with providing minimum 2 days - solids retention time at 30 ˚C to achieve %10 VFAs/gVSS production yield. As it could be seen from the Figure 8.4, 3,000 m^3/d primary sludge with a DS content of produces in the WWTP. The volatile part of this sludge is nearly 60%. From this knowledge, it could be estimated that 36 tons of volatile solids produce in the PS. Assuming VFAs generation yield as 10%, it could be concluded that 3.6 tonVFAs/d production rate cold be achieved. This amount of VFAs will be sufficient to remove 0.9 tons P/d. Therefore 70% percent of additional phosphorus removal requirement could be provided with prefermentation.

Especially if this alternative were combined together with elimination of phosphorus releases from sludge handling units, effluent phosphorus target could be achieved.

However several modifications in the primary sludge collection and pumping systems is necessary to obtain a stable system. These revisions may include, automatic primary sludge suction system controlling with online turbidity sensors allowing to suck primary sludge with a desired DS content; modified bottom scrapers; valves and sludge heating systems etc. In addition to these modifications, since some part of the organic material is acidified in fermentation reactions the quantity of the methane gas from anaerobic digestion process decreases. During the process high concentrations of odor compounds could be released. Therefore adequate odor control system is required.

8.2.3 Optimizing operational conditions to improve EBPR efficiency

Efficiency of the EBPR processes is hindered behind domination of PAOs with in the activated sludge system. However, it is evident from Figure 6.7 that even if PAOs dominated with in the system, their performance is still dependent upon operational conditions including hydraulic retention time, sludge age, internal and external recirculation.

Lötter and Murphy (1985) reported that *Pseudomonas*, *Aeromonas* and *Acinetobacter* accomplished denitrification in the anoxic stage of a biological nitrogen removal system. Similar results were also reported by Jorgensen and Pauli (1995) that the denitrifiers, such as *Pseudomonas, Hydrogenophaga, Citrobacter, Xanthomonas*, have the ability of phosphorus accumulation. The excess phosphorus uptake rates of these bacteria were higher than those reported for *Acinetobacter strains*. These results showed that polyphosphate accumulation and denitrification in activated sludge can be carried out by the same microorganisms. It has also been proposed that *Pseudomonas* could actively take part in simultaneous denitrification and phosphorus uptake in EBPR sludge (Atkinson *et al*., 2001).

Furthermore, since PAOs obtain more energy as compared to anaerobic ATP conversion reactions, PAOs would prefer to obtain energy from denitrification reactions in the anaerobic stage of the EBPR processes using electron acceptors present in influent and return sludge streams. Consequently, well-known steps of anaerobic EBPR reactions mainly including release of phosphorus and storage of VFAs as PHA in the cell inventory could not happen until all electron acceptors are depleted.

This study revealed that anaerobic retention time (HRT_a) is one of the main operational parameters affecting many fundamental characteristics EBPR process. Several investigators developed EBPR models based on COD storage in anaerobic tank resulting with release of phosphorus. It could be safely evaluated that insufficient HRT_a is going to be caused insufficient uptake of VFAs that is stored as PHB and release of phosphorus consequently. Therefore this short contact period (1-2 hours) will influence all of the remaining oxidative biochemical reactions lasting up to 7 hours. These oxidative EBPR reactions includes anoxic and aerobic uptake of phosphorus in which required energy is derived from stored PHA in the anaerobic zone. Therefore domination of PAOs is directly dependent upon HRT_a. Laboratory scale batch tests indicated that anaerobic reactions last 90 minutes. As it could be seen form Figure 8.8, phosphorus release and COD utilization last minimum 90 minutes. In the full scale experiments it was also determined that HRT_a should be longer than 1.1 hours.

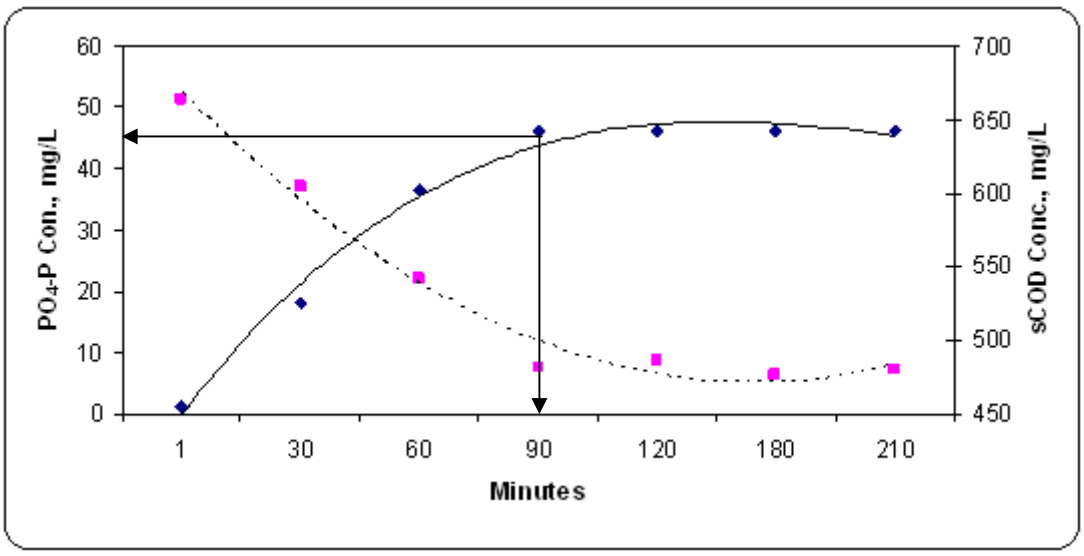


Figure 8.8 Anaerobic reaction profile of a stabile EBPR process

Previously investigations indicated that presence of electron acceptors in the anaerobic zone has a net reducing effect on influent carbon source. In this study it is also found that electron acceptors could also influence available anaerobic retention time (AART) for PAOs.

Consequents of this new finding is every important for full scale WWTPs especially designed with short HRT_a (1 hours). As it could be seen from the Figure 8.9, AART that is required for completion of biochemical EBPR reactions will be significantly decreased with lowering specific denitrification rate (q_{dn}). It was well established by the previous studies that denitrification reactions were significantly affected by temperature, pH etc. environmental conditions (USEPA, 1993; Metcalf & Eddy, 2003). Especially, low wastewater temperature could cause significant deterioration in denitrification reaction rate and so complete cessation of EBPR reactions may occur.

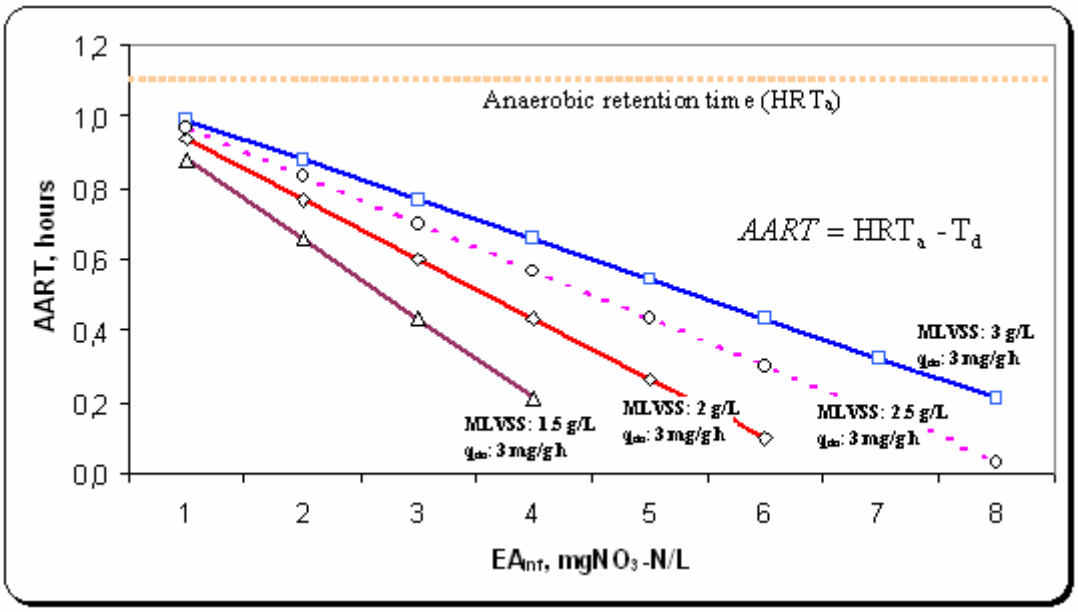


Figure 8.9 Available anaerobic reaction time (AART) as a function of EA & MLVSS concentration

These findings clearly indicated that sufficient anaerobic retention time should be provided for the completion of all EBPR reactions and utilization of electron acceptors via denitrification reactions. The volume of anaerobic tanks should also respond daily peak flow variations.

Sludge age is another identical operational parameter that could affect the EBPR reactions. General concept for sludge age control should be established together with temperature. It has been reported that phosphorus removal performance was optimum at SRT ranges of 16-24 days and 12 to 17 days for 5 and 10 °C. Higher SRT values up to 32 days at 5 °C and 25 days at 10 °C reduced EEBPR performance. This deterioration was explained by increased extent of endogenous respiration which consumed internally stored glycogen, leaving less reducing power of PHA formation in anaerobic stages. The washout SRT of each system found as 3.5 days at 5 °C and 1.8 days 5 °C (Erdal (U.G.), Erdal (Z.K.), Randal, 2003). During the full scale investigations sludge age varied between 7 days to 20 days and EBPR efficiency up to 88% were recorded.

Internal and external sludge recirculation is also important not only for EBPR efficiency but also important for allover efficiency of the biological treatment steps including carbon and nitrogen removal. According to the Table 2.6, return activated sludge (RAS) ratio based on inflow may vary from 25% to 100% according to the process type. Moreover, WWTPs configured for combined removal of nitrogen and phosphorus has RAS percentages changing between 50 and 100. It could be recommended that RAS should guarantee sufficient degree of MLVSS in the biological reactors. Internal recirculation is necessary to transfer nitrate-rich mixed liquor from aerobic zone to anoxic zone for complete biological denitrification. The internal recirculation rate based on inflow may vary between 100% and 400% according to the process type (Metcalf & Eddy 2003). In this study, internal recirculation rate were constant as 400% of inflow and during the monitoring period nitrogen removal efficiency were always higher than 70% with an average of 80%.

Dissolved oxygen (DO) control in EBPR plants is one of other critical operational concerns. Control of DO is important for optimization of both EBPR efficiency and operational costs. Long term operational experiences indicated that energy cost is the biggest cost item in WWTPs. As discussed in the previously chapters EBPR processes consist of several different oxidative environment including anoxic and aerobic zones. It is well documented in the literature that DO concentration in the

aerobic zone should be higher than 2 mg/L. However in the anoxic zone, DO concentration should be less that 0.5 mg/L (Metcalf & Eddy, 2003). Therefore reliable oxygen control systems should be used in the EBPR plants. These systems should be based on online DO measurement probes and air flow given to the aeration tanks should be adjustable according to the variable influent organic loading rates. In fact, while insufficient DO levels may cause several process deteriorations including decreased nitrification efficiency, stimulation of filamentous bacteria etc., excess amounts of DO will increase the operational costs. Another concern is inhibition of denitrification reactions as a result of DO concentration higher than 1 mg/L (Metcalf & Eddy, 2003).

Final clarifiers of the EBPR systems should be well-designed to minimize suspended solid lost to the effluent. These facilities should overcome changeable surface loading rates. Results of this study clearly indicated that MLVSS phosphorus content could increase up to 10%. This means that loss of these suspended solids will increase the effluent total phosphorus concentration. One of the disadvantages of the EBPR plats with weak wastewater organic content is occurrence of high SVI values. During the monitoring period, recorded values indicated that SVI could increase up to 200 ml/g. According to the literature optimum SVI value should be with in the range of 100 – 150 ml/g. Therefore, design and operation of final clarifiers is very important concern in WWTPs.

Figure 8.10 was adapted from ATV Standard, ATV-DVWK-A 131E, published on May 2000 and this figure could provide more reliable control for minimize solids losses from final clarifiers. According to the standard, a diluted sludge volume (DSV) is defined as a function of SVI and could be expressed as following equation;

$$DSV(l/m^3) = SVI \times MLSS$$

In this equation, SVI is defined as sludge volume index, l/kg and MLSS is defined as mixed liquor suspended solids concentration as kg/m^3.

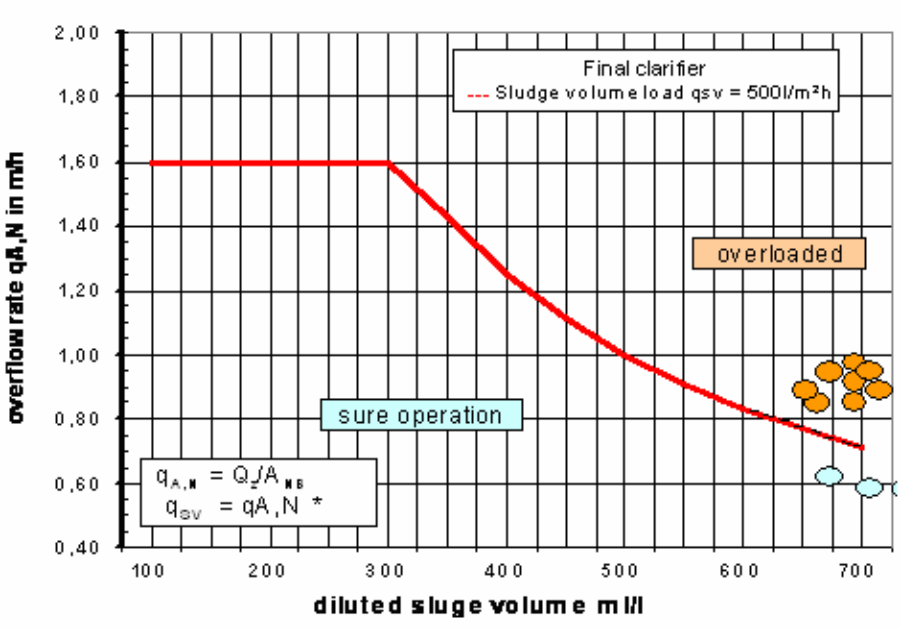


Figure 8.10 Diluted sludge volume as a function of overflow rate

Over flow rate ($q_{A,B}$) is estimated by dividing flow rate (m^3/h) to surface area of final clarifier (m^2). Then sludge loading rate (q_{SV}) is estimated by multiplying over flow rate by DSV. This obtained sludge loading rate ($l/m^2.h$), should be lower than 450 ($l/m^2.h$). Figure 8.10 could also be used to prevent overload of final clarifiers and to keep them in sure operation limits. (Estimated sludge loading rate should be kept under the red marked line depicted in Figure 8.10).

Management strategies for improvement of phosphorus removal in WWTPs were discussed through the above given sections. Findings of this thesis clearly indicated that effluent phosphorus level could be controlled by EBPR processes with significant efficiencies up to 90%. However investigation results also indicated that EBPR processes are highly dependent upon wastewater characteristics and operational conditions of the WWTP. Especially organic composition of influent was directly effective on establish a stable EBPR performance. Therefore, both at design and operation levels, wastewater should be characterized accurately for nutrients and their various forms. Especially all fractions of COD, nitrogen and phosphorus should be well defined to establish required treatment stages. At design level, to decide if wastewater characteristics are ideal for EBPR processes, batch scale tests described in chapter 3.2, could be applied practically. At existing wastewater treatment

facilities configured with EBPR configuration, mass balances should be established in biological treatment units and sludge handling facilities to improve the process efficiency.

Control of effluent phosphorus level by chemical precipitation method is also one of the common methods in worldwide. However, revised literature sources indicated that chemical phosphorus precipitation could cause significant increase in sludge production rate in WWTPs. It was also reported that characteristics of these chemical sludges were completely different as compared to the non-chemical sludges. In several cases, unit sludge operations (thickening and dewatering) and final sludge disposal applications (thermal drying and incineration etc.) could be adversely affected. It could also be safely reported that increased sludge amounts and higher sludge moisture content will increase the operational costs and required maintenance works for sludge handling facilities. As it was discussed in the previous section, EBPR processes are influenced by many wastewater and operational variables and process stability could not be maintained annually. Although chemical phosphorus removal methods could cause several problems, mainly associated with sludge handling, combination of EBPR process with chemical phosphorus precipitation method could be helpful to achieve strict effluent discharge limits. However, if these two methods are considered in the general flow scheme several critical concerns should be taken into consideration. The most important concern is competition between metal salt and PAOs. It was demonstrated in all chapters of this thesis that phosphate ion is vital for PAOs in all biochemical reactions. Required energy for substrate utilization, reproduction and cell maintenance is supplied from phosphate ion. Therefore addition of excess amounts of metal ion could remove available phosphate from aqueous phase. In this case PAOs will definitely become out of competition with ordinary heterotrophic bacteria. This will also cause an increase in the required metal salt dosage. Therefore, if use of chemicals were planned for phosphorus precipitation, optimum dosage should be determined by continuous online measurement systems. Moreover, metal salt addition could be performed at the end of the aeration tank before the final clarifiers for a safer operation. In this case, sufficient degree of velocity gradient should be provided.

CHAPTER NINE
CONCLUSIONS

9.1 Conclusions

In this study, it was aimed to investigate management of phosphorus removal in full scale WWTP. Both full scale and batch scale methods were performed to trace phosphorus recirculation in allover the treatment plant facilities. Effect of variable hydraulic and organic loading rates on fundamental characteristics of EBPR process were investigated deeply even in microbial intracellular environment. Findings of the study were supported by literature to establish a reliable phosphorus management strategy in WWTPs.

Performance of the EBPR process was investigated under different anaerobic hydraulic retention times in combination with electron acceptor input to the anaerobic zone. It was found that EBPR efficiency could be highly dependent upon AART. Recorded TP removal efficiencies ranged from 49% to 80% while AART increasing from 0.76 h to 1.1 h. Activated sludge P content was 5.1% for 0.76 h AART and 10.1% for 1.1 h AART. It was also observed that AART higher than 1 hour resulted with increase of PAOs intercellular P content up to 30%. Since PAOs obtain more energy as compared to anaerobic ATP conversion reactions, PAOs would prefer to obtain energy from denitrification reactions in the anaerobic stage of the EBPR processes using electron acceptors present in influent and return sludge streams. This study revealed that anaerobic retention time (HRT_a) is one of the main operational parameters affecting many fundamental characteristics EBPR process. Several investigators developed EBPR models based on COD storage in anaerobic tank resulting with release of phosphorus. It could be safely evaluated that insufficient HRT_a could cause insufficient uptake of VFAs that could lead a decreased PAOs mass fraction in the system. Therefore this short contact period could influence all of the remaining oxidative biochemical reactions including carbon and nitrogen removal. These results were validated by laboratory scale batch tests. These tests clearly showed that anaerobic reaction could last up to 1.5 hours.

Another adverse effect of short anaerobic retention time may be observed in the following anoxic stage of the EBPR processes. Soluble external carbon sources remaining from the anaerobic stage could enter to the anoxic zone. Long-term operational experiences indicated that presence of external carbon sources in the anoxic zone of EBPR process may deteriorate anoxic P uptake and stimulate filamentous and ordinary heterotrophic bacterial growth in combination with low temperature.

The results of full scale investigation clearly showed that rbsCOD/TP ratio could control EBPR process. Short-term effects of COD limited conditions could cause slower COD utilization and P release rates in the anaerobic zone and continues with deteriorated P uptake in aerobic zone and anoxic zone. Measurement results proposed that gradually increase of rbsCOD/TP ratio from 6 to 21, could improve the EBPR efficiency up to 88%. P content of the activated sludge may also increase to 10.5% parallel to increase of rbsCOD/TP ratio. Obtained results also proposed that PAOs could not exist in the activated sludge in large quantities under COD limited conditions. Furthermore, rbsCOD/TP ratio could be used as reliable criteria for design and operation of EBPR processes. Long term wastewater characterization results indicated that wastewater of İzmir city could be COD limited (rbsCOD/TP<10) in the summer season. Therefore regulatory cautions could be developed to improve influent carbonaceous level. Industrial wastewaters containing high amounts of readily biodegradable carbon sources could be discharged to the sewerage system without pre treatment.

Activated sludge characterization results clearly indicated that without sufficient rbsCOD/TP ratio, PAOs could not be selected within the process although all important EBPR parameters within the acceptable limits. Increase of PAOs mass fraction up to 30% could provide with significantly higher EBPR efficiency (88%). However insufficient HRT_a could deteriorate EBPR efficiency even though there are sufficient quantities of PAOs. Since presence of electron acceptors have a net reducing effect on soluble substrate forms, domination of PAOs could also be limited by the mechanism both decreasing rbsCOD/TP ratio and decreasing anaerobic

available reaction time. Intracellular P content of the PAOs varied from 10% to 30% and mass fraction of PAOs was within the range of 9% to 34% during the whole monitoring period. Observed main difference between previously conducted laboratory scaled tests and results of this full scale study was significantly higher mass fraction of OHOs in the activated sludge culture. Average mass fraction of these microorganisms was 70%. This gradually higher OHOs existence could be explained with particulate fraction of influent COD. Moreover, electron acceptor input to the anaerobic zone and fermentation degree of rbsCOD could control mass fraction of PAOs with in the process.

Measurement results indicated that 3,000 m^3/d primary sludge with a DS content of 2% is produced in the WWTP. Using previously reported VFAs generation rate, 3.6 ton VFAs / d production rate could be achievable in the WWTP. This amount of VFAs will be sufficient to remove 900 kg P/d. Therefore 70% percent of additional phosphorus removal requirement could be provided with prefermentation technique. Especially if this alternative were combined together with elimination of phosphorus releases from sludge handling units, effluent phosphorus target could be achieved. However several modifications in the primary sludge collection and pumping systems is necessary to obtain a stable system. These revisions may include, automatic primary sludge suction system controlling with online turbidity sensors allowing to suck primary sludge with a desired DS content; modified bottom scrapers; valves and sludge heating systems etc. In addition to these modifications, since some part of the organic material is acidified in fermentation reactions the quantity of the methane gas from anaerobic digestion process decreases. During the process high concentrations of odor compounds could be released. Therefore adequate odor control system is required.

COD/TP ratio could be optimized eliminating undesired phosphorus releases from the sludge handling facilities of the WWTPs. According to the mass balance results performed around sludge equalization tanks and sludge dewatering system, significant amount of phosphorus is released and reloaded back to the biological treatment units. Estimations clearly indicated that elimination of these secondary

releases could provide a higher COD/TP ratio. This modification could result with additional, 30% P removal improvement consequently.

It was found that minimum influent VFAs concentration is 47 mg HAC/l that corresponds 29 ton HAC/d loading rate. In addition, 2 tons/d of electron acceptor (as nitrate equivalent) inputs to the anaerobic tanks and as result of that, 10 ton/d VFAs is utilized in the denitrification reactions. Therefore 19 ton/d VFAs is available for the PAOs that suffice removal of 4.7 ton daily P load. Moreover, nearly 34% percent of simple substrate forms (VFAs) that is vital for proliferation of PAOs could utilized in undesired denitrification reactions. Therefore elimination of electron acceptor input to the anaerobic zone is very critical concern for full scale WWTPs.

Full scale investigations indicated that dissolved oxygen is one of the important electron acceptors and mainly carried to the anaerobic tank by both influent and return sludge. It could be safely concluded that high velocity gradients and turbulence caused oxygenation of both influent and return sludge streams. Therefore hydraulic design of return sludge transfer system and wastewater distribution chamber existing on the head of anaerobic tank should be well-designed to eliminate turbulences in both side and main streams.

Nitrate is another significant electron acceptor and measurement results indicated that concentration of nitrate in the return sludge line could increase up to 4 mg/L. To eliminate nitrate entrance to the anaerobic tank, post denitrification zone should be well designed and should have sufficient volume to provide enough contact time for nitrate utilization in which carbon are derived from endogenous respiration.

It was demonstrated by both full-scale and batch-scale investigations that biodegradable particulate substrate forms could enhance the performance of EBPR and sludge settling characteristics. Ability of the activated sludge to utilize COD with the aid of P release in the anaerobic zone, anoxic and aerobic P uptake rates and denitrification rate could be increased up to 60%. In addition to faster EBPR biochemical reaction rates, successfully selection of the PAOs with in the activated

sludge system could increase the denitrification reactions up to 75% which also means increased nitrogen removal efficiency.

In addition to wastewater characteristics, operational variables may also influence the EBPR performance. Sludge age is one the important operational parameters. General concept for sludge age control could be based on activated sludge temperature. Although this study has not special experimental design to determine optimum sludge age, EBPR efficiency of 88% could be achievable at sludge age range of 7- 20 days and temperature range of 15 – 25 ˚C. For a generalized management concept, longer sludge ages should be maintained in low temperatures to establish a functioning EBPR process. However, too long sludge age values could results with endogenous respiration that could deteriorate the EBPR efficiency due to secondary P releases.

It was also found that dissolved oxygen control in EBPR plants is one of other critical operational concerns. Control of DO is important for optimization of both EBPR efficiency and operational costs. Long term operational experiences indicated that energy cost is the biggest cost item in WWTPs. It is well documented in the literature that DO concentration in the aerobic zone should be higher than 2 mg/L. However in the anoxic zone, DO concentration should be less that 0.5 mg/L. Therefore a reliable oxygen control systems should be used in the EBPR plants. These systems should be based on online DO measurement probes and air flow given to the aeration tanks should be adjustable according to the variable influent organic loading rates. In fact, while insufficient DO levels may cause several process deteriorations including decreased nitrification efficiency, stimulation of filamentous bacteria etc., excess amounts of DO will increase the operational costs. Another concern is inhibition of denitrification reactions as a result of DO concentration higher than 1 mg/L.

Another important finding of this study was related to the final clarifier design and operation. Final clarifiers of the EBPR systems should be well-designed to minimize suspended solid lost to the effluent. These facilities should overcome changeable surface loading rates. Full scale measurement results indicated that MLVSS phosphorus content of the activated sludge could increase up to 10%. This means that loss of these suspended solids will increase the effluent total phosphorus concentration significantly. One of the disadvantages of the EBPR plants loading with weak wastewater organic content is occurrence of high SVI values. During the monitoring period, recorded values indicated that SVI could increase up to 200 ml/g. According to the literature optimum SVI value should be with in the range of 100 – 150 ml/g. Therefore, design and operation of final clarifiers is very important concern in WWTPs.

Findings of this thesis clearly indicated that effluent phosphorus level could be controlled by EBPR processes with significant efficiencies up to 90%. However investigation results also proposed that EBPR processes are highly dependent upon wastewater characteristics and operational conditions of the WWTP. Therefore EBPR processes could be supported with chemical phosphorus precipitation methods. However, revised literature sources clearly indicated that chemical phosphorus precipitation could not be used alone to achieve effluent phosphorus target and also they could cause significant increase in sludge production rate in WWTPs. It was also reported that characteristics of these chemical sludges were completely different as compared to the non-chemical sludges. In several cases, unit sludge operations (thickening and dewatering) and final sludge disposal applications (thermal drying and incineration etc.) could be adversely affected. It could also be safely concluded that increased sludge amounts and higher adhered sludge moisture content will increase the operational costs and required maintenance works for sludge handling facilities. Although chemical phosphorus removal methods could cause several problems combination of EBPR process with chemical phosphorus precipitation method could be helpful to achieve strict effluent discharge limits. However, if these two methods are considered in the general flow scheme several critical precautions should be taken into consideration. The most important concern

is competition between metal salt and PAOs. It was demonstrated in all chapters of this thesis that phosphate ion is vital for natural selection of PAOs within the activated sludge system. Required energy for substrate utilization, reproduction and cell maintenance is supplied from phosphate ion. Therefore addition of excess amounts of metal ion could remove available phosphate from aqueous phase. In this case PAOs will definitely become out of competition with ordinary heterotrophic bacteria. This will also cause an increase in the required metal salt dosage. Therefore, if use of chemicals were planned for phosphorus precipitation, optimum dosage should be determined by continuous online measurement systems. Moreover, metal salt addition could be performed at the end of the aeration tank before the final clarifiers for a safer operation. In this case, sufficient degree of velocity gradient should be provided.

9.2 Suggestions

Full-scale investigations could be performed using online nutrient measurement systems to obtain a continuous monitoring period that would give more detailed investigation results. This method could also eliminate possible sampling problems; provide more accurate analysis results especially if the sampling area is large. Monitoring of influent TOC parameter by online measurement systems could also provide a detailed database for a EBPR process.

In this study importance of biodegradable suspended materials in EBPR was clearly demonstrated and proved. However, the mechanism of particulate COD removal through out the pathways of EBPR could be investigated deeply.

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Nomenclature

Q_i	Inflow, m^3/s
Q_r	Recycle flow, m^3/s
Q_{int}	Internal recycle flow, m^3/s
Q_w	Waste activated sludge flow, m^3/d
ΣQ	Total flow, m^3/d
$V_{anaerobic}$	Anaerobic tank volume, m^3
HRT	Hydraulic retention time, hour
HRT_a	Anaerobic hydraulic retention time, hour
$[PO4\text{-}P_{ad}]$	PO4-P concentration after digestion, mg/L
$[PO4\text{-}P_{af}]$	PO4-P concentration after filtration, mg/L
Sf	Balanced influent nutrient concentration, mg/L
S	Influent nutrient concentration
S_r	Return sludge nutrient concentration
$[\Sigma PO_4\text{-}P_{anx\text{-}inf}]$	Balanced PO_4-P concentration in anoxic zone, mg/L
$[PO_4\text{-}P_{anx\text{-}inf}]$	PO_4-P concentration in the anaerobic tank effluent, mg/L
$[PO_4\text{-}P_{ae}]$	PO_4-P concentration in the aerobic zone, mg/L
NO	Influent total nitrogen, mg/L
NH_e	Effluent ammonium concentration, mg/L
N_{syn}	Nitrogen used in solids synthesis, mg/L
Y_n	Yield coefficient, gVSS/gCOD
F_n	Fraction of nitrogen in MLVSS, %
P_x	Cell Synthesis
F_p	Phosphorus Content of Mixed Liquor Suspended Solids
$[NO_3\text{-}N_{removed}]$	Removed nitrate concentration in anoxic zone, mg/L
$[NO_3\text{-}N_{eff}]$ is	Effluent nitrate concentration, mg/L
$Px_{PAOs\text{-}anoxic}$	Biomass production as a result of nitrate removal, $gVSS/m^3$
Y_{dn}	Denitrifier synthesis yield, gVSS/d NO_3-N
$[PO_4\text{-}P_{rem\text{-}anoxic}]$	Removed concentration of PO_4-P in anoxic zone, mg/L
$\Sigma[PO_4\text{-}P_{anoxic\text{-}inf}]$	Balanced concentration of PO_4-P in anoxic zone, mg/L
$[PO_4\text{-}P_{eff}]$	Effluent PO_4-P concentration, mg/L
ΣPx	Total biomass production, $gVSS/m^3$
ΣPx_{PAOs}	Biomass production from rbsCOD utilization (PAOs cell synthesis), $gVSS/m^3$
$\Sigma Px_{non\text{-}PAOs}$	Biomass production from bpCOD utilization (non - PAOs cell synthesis), $gVSS/m^3$
Y_h	Heterotrophic bacteria synthesis yield, gVSS/gCOD

k_d	Endogenous decay coefficient, gVSS/gVSS.day
SRT_{ae}	Aerobic solids retention time, days
X_e	Concentration of biomass in effluent, g/m^3
X_R	MLVSS concentration of return sludge, g/m^3
ΣbCOD	Balanced concentration of influent total biodegradable COD, mg/L
ΣrbsCOD	Balanced concentration of influent total soluble readily biodegradable COD, mg/L
f_d	Fraction of cell mass remaining as cell debris, g/g
$\Sigma PO_4\text{-}P_{anoxic}$	Measured PO_4-P concentration in anoxic zone, mg/L
$PO_4\text{-}P_{eff}$	PO_4-P concentration in the effluent, mg/L
$PO_4\text{-}P_{rem\text{-}part}$	Concentration of PO_4-P removed during particulate COD utilization, mg/L
$[sCOD]_{net}$	sCOD used in P release, mg/L
$[sCOD]_t$	Total amount of sCOD used during anaerobic period, mg/L
$[sCOD]_d$	sCOD used in denitrification reactions, mg/L
q_{sCOD}	Anaerobic sCOD utilization rate, mgsCOD/gVSS/h
$[PO_4\text{-}P]_r$	Net amount of P release, mg/L
$[PO_4\text{-}P]_e$ is	Measured amount of P in the effluent of anaerobic tank, mg/L
$\Sigma[PO_4\text{-}P]_i$	Balanced concentration of P in the influent of anaerobic tank, mg/L
$[PO_4\text{-}P]_u$	Net amount of P uptake, mg/L
$[MLVSS]_{aerobic}$	Mixed liquor volatile suspended solids concentration in aerobic zone, g/m^3
HRT_{ae}	Hydraulic retention time in the aeration tank, hours
PHA	Poly Hydroxyl Alkanoate
PHB	Polyhydroxybutyrate
TN	Total Nitrogen
SRT	Solids Retention Time
θ_c	Sludge Age
r^2	Statistical Correlation Coefficient
WWTP	Wastewater Treatment Plant
PLC	Programmable Local Control

Abbreviations

SS	Suspended solids, mg/L
VSS	Volatile suspended solids, mg/L
MLSS	Mixed liquor suspended solids, mg/L
MLVSS	Mixed liquor volatile suspended solids, mg/L
BOD	Biochemical oxygen demand, mg/L
UBOD	Ultimate biochemical oxygen demand, mg/L
COD	Chemical oxygen demand, mg/L
sCOD	Soluble chemical oxygen demand, mg/L
bCOD	Biodegradable COD, mg/L
pbCOD	Particulate biodegradable COD, mg/L
rbsCOD	Readily biodegradable soluble chemical oxygen demand, mg/L
nbsCOD	Non Biodegradable Soluble Chemical Oxygen Demand, mg/L
TP	Total Phosphorus, mg/L
TN	Total Nitrogen, mg/L
EBPR	Enhanced Biological Phosphorus Removal
PAOs	Phosphorus Accumulating Microorganisms
GAOs	Glycogen Accumulating Microorganisms
DO	Dissolved Oxygen
AO	Anaerobic / Oxic
A^2O	Anaerobic / Anoxic / Oxic
UCT	University of Cape Town
rxl	Recycle
PHA	Poly Hydroxyl Alkanoate
PHB	Polyhdroxybutyrate
SRT	Solids Retention Time
PS	Primary Sedimentation
RAS	Return Activated Sludge, %
WWTP	Wastewater Treatment Plant
PLC	Programmable Local Control

APPENDECIES

APPENDIX- A

Raw Data for Full-Scale Investigations (First Part)

Table A.1. Full-scale experimental results conducted on 10.01.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L	
10.01.2006	CLOUDY	I (PS Online)	Influent	NM	NM	NM	38	35,0	ND	9,1	NM	NM	
			Bio-P_{inf}	245	780	220	38	22	ND	7,5	5,4	1,8	
			Bio-P_{eff}	NM	NM	40	NM	NM	ND	55	10,5	ND	
			Anoxic	NM	NM	35	NM	NM	2,6	NM	6,5	0,3	
			Aerobic	NM	NM	40	NM	NM	3,5	138	2,8	2,2	
			Effluent	NM	50	30	10	0,5	2,6	3,4	2,5	3,5	
			Recycle	NM	NM	25	NM	NM	2,5	NM	2,0	2,0	
Date	**Sampling point**	**Q_{inf} m^3/s**	**Q_{es} m^3/s**	**Q_{rxl} m^3/s**	**Q_{int} m^3/s**	**MLSS mg/L**	**LOI %**	**MLVSS mg/L**	**Mixed Sludge P Content %**		**Mixed Sludge N Content %**	**SVI ml/g**	**TEMP. ^oC**
10.01.2006	Anaerobic	1,8	2385	1,8	2,72	3500	72	2520	6,2		6%	150	16,5
	Anoxic					3400	70	2380					
	Aerobic					3100	70	2170					
	Recycle					8800	74	6512					
	Excess					8800	74	6512					

Table A.2. Full-scale experimental results conducted on 28.01.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
20.01.2006	CLOUDY	I (PS Online)	Influent	NM	720	NM	NM	38,0	ND	9,1	NM	NM
			Bio-P_{inf}	242	710	208	48	NM	ND	8,1	5,8	2,5
			Bio-P_{eff}	NM	NM	40	NM	NM	ND	53	11,6	ND
			Anoxic	NM	NM	55	NM	NM	1,6	NM	6,5	0,5
			Aerobic	NM	NM	55	NM	NM	6	121	3,5	2,1
			Effluent	NM	60	40	12	1	1,5	3,9	2,9	3,2
			Recycle	NM	NM	40	NM	NM	1,6	NM	2,7	1,7

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
20.01.2006	Anaerobic	2,2	1900	1,8	2,72	3300	70	2310	5,6	6%	140	15,5
	Anoxic					3200	70	2240				
	Aerobic					3100	68	2108				
	Recycle					7500	73	5475				
	Excess					7500	73	5475				

Table A.3. Full-scale experimental results conducted on 14.02.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
14.02.2006	Sunny	I (PS Online)	Influent	NM	500	NM	NM	NM	ND	7,1	NM	NM
			Bio-P_{inf}	147	430	154	35	NM	ND	6,5	4,5	2,1
			Bio-P_{eff}	NM	NM	40	NM	NM	ND	50	11,1	ND
			Anoxic	NM	NM	36	NM	NM	0,6	NM	3,7	0,5
			Aerobic	NM	NM	36	NM	NM	3,6	175	0,4	2,1
			Effluent	NM	60	40	5,1	0,5	0,25	1,3	0,6	3,2
			Recycle	NM	NM	36	NM	NM	0,5	NM	0,6	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
14.02.2006	Anaerobic	1,7	3518	1,8	2,72	3100	65	2015	10,2	6%	150	15,0
	Anoxic					2900	66	1914				
	Aerobic					2600	66	1716				
	Recycle					8000	70	5600				
	Excess					8000	70	5600				

Table A.4. Full-scale experimental results conducted on 22.02.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
22.02.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	45	NM	ND	NM	NM	NM
			Bio-P_{inf}	198	500	185	38	22,3	ND	9,5	5,4	2,5
			Bio-P_{eff}	NM	NM	55	NM	NM	ND	60	9,2	ND
			Anoxic	NM	NM	36	NM	NM	3,6	NM	6,2	0,5
			Aerobic	NM	NM	42	NM	NM	5,5	148	3,6	2,1
			Effluent	NM	70	38	11	3,5	3,1	4,2	3,6	3,2
			Recycle	NM	NM	40	NM	NM	3,4	NM	3,5	1,5

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
22.02.2006	Anaerobic	2,8	2530	1,8	2,72	3600	70	2520	6,1	NM	160	16
	Anoxic					3500	70	2450				
	Aerobic					3400	70	2380				
	Recycle					6200	70	4340				
	Excess					6200	70	4340				

Table A.5. Full-scale experimental results conducted on 28.02.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
28.02.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	45	NM	ND	NM	NM	NM
			Bio-P_{inf}	167	440	170	35	18,6	ND	7	4,2	2,1
			Bio-P_{eff}	NM	NM	35	NM	NM	ND	55	11,6	ND
			Anoxic	NM	NM	36	NM	NM	1,5	NM	4,5	0,5
			Aerobic	NM	NM	34	NM	NM	4	220	0,8	2,1
			Effluent	NM	45	20	5,5	0,2	1,4	1,4	0,6	3,2
			Recycle	NM	NM	38	NM	NM	1,5	NM	0,6	1,5

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
28.02.2006	Anaerobic	1,8	2754	1,8	2,72	3500	70	2450	10,1	6%	140	16
	Anoxic					3300	70	2310				
	Aerobic					3200	68	2176				
	Recycle					7500	72	5400				
	Excess					7500	72	5400				

Table A.6. Full-scale experimental results conducted on 02.03.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
02.03.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	NM	NM	ND	NM	NM	NM
			Bio-P_{inf}	167	420	180	40	25,3	ND	9	5,0	2,1
			Bio-P_{eff}	NM	NM	45	NM	NM	ND	60	11,2	ND
			Anoxic	NM	NM	30	NM	NM	1,4	NM	4,0	0,5
			Aerobic	NM	NM	45	NM	NM	3,3	214	1,2	2,1
			Effluent	NM	35	25	7	1,2	1,5	1,8	1,0	3,2
			Recycle	NM	NM	33	NM	NM	1,8	NM	0,6	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
02.03.2006	Anaerobic	1,8	2900	1,8	2,72	3500	70	2450	10,1	NM	140	16
	Anoxic					3300	70	2310				
	Aerobic					3000	70	2100				
	Recycle					8200	72	5904				
	Excess					8200	72	5904				

Table A.7. Full-scale experimental results conducted on 14.03.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
14.03.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	50	NM	NM	12,28	7,0	-
			Bio-P_{inf}	114	290	130	32	22,6	ND	9,5	6,2	2,8
			Bio-P_{eff}	NM	NM	45	NM	NM	ND	48	11,0	ND
			Anoxic	NM	NM	40	NM	NM	3,5	NM	6,6	0,5
			Aerobic	NM	NM	40	NM	NM	7,6	88	5,0	2,1
			Effluent	NM	50	42	10,5	1,4	3,8	4,8	4,3	3,2
			Recycle	NM	NM	50	NM	NM	3,6	NM	3,9	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
14.03.2006	Anaerobic	2,8	3360	1,8	2,72	2850	65	1852,5	5,1	NM	140	15
	Anoxic					2600	64	1664				
	Aerobic					2500	65	1625				
	Recycle					8600	70	6020				
	Excess					8600	70	6020				

Table A.8. Full-scale experimental results conducted on 02.04.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
02.04.2006	Sunny	I (PS Online)	Influent	NM	450	NM	40	28,0	NM	NM	NM	-
			Bio-P_{inf}	138	350	160	34	NM	ND	8,5	5,1	2,5
			Bio-P_{eff}	NM	NM	45	NM	NM	ND	70	10,5	ND
			Anoxic	NM	NM	NM	NM	NM	0	NM	4,0	0,5
			Aerobic	NM	NM	40	NM	NM	5,4	281	1,8	2,1
			Effluent	NM	55	45	6,5	0,4	2,2	2,1	2,0	3,2
			Recycle	NM	NM	38	NM	NM	2,3	NM	1,7	1,5

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. °C
02.04.2006	Anaerobic	2,3	2380	1,8	2,72	4200	75	3150	9,4	6%	130	19
	Anoxic					4100	75	3075				
	Aerobic					4000	74	2960				
	Recycle					11500	76	8740				
	Excess					11500	76	8740				

Table A.9. Full-scale experimental results conducted on 21.04.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
21.04.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	30	NM	NM	NM	NM	-
			Bio-P_{inf}	103	300	112	25	16,5	ND	5,5	3,9	1,6
			Bio-P_{eff}	NM	NM	30	NM	NM	ND	50	7,7	ND
			Anoxic	NM	NM	42	NM	NM	0,8	NM	3,3	0,5
			Aerobic	NM	NM	NM	NM	NM	2,2	190	0,6	2,1
			Effluent	NM	40	38	3,8	3	0,55	1,2	1,0	3,2
			Recycle	NM	NM	42	NM	NM	0,6	NM	0,7	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
21.04.2006	Anaerobic	1,8	2380	1,8	2,72	2900	70	2030	9,6	6%	110	21
	Anoxic					2800	70	1960				
	Aerobic					2780	71	1973,8				
	Recycle					6800	72	4896				
	Excess					6800	72	4896				

Table A.10. Full-scale experimental results conducted on 01.05.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
01.05.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	30	NM	NM	NM	NM	-
			Bio-P_{inf}	153	380	170	36	24,8	ND	11,1	5,5	2,4
			Bio-P_{eff}	NM	NM	70	NM	NM	ND	60	9,5	ND
			Anoxic	NM	NM	40	NM	NM	1,2	NM	5,9	0,5
			Aerobic	NM	NM	NM	NM	NM	6,3	170	4,4	2,1
			Effluent	NM	50	42	12	3,5	1,3	4,4	4,1	3,2
			Recycle	NM	NM	41	NM	NM	1,5	NM	3,8	1,9

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
01.05.2006	Anaerobic	2,8	2400	1,8	2,72	3740	70	2618	6,5	NM	90	20
	Anoxic					3600	71	2556				
	Aerobic					3580	71	2541,8				
	Recycle					9100	73	6643				
	Excess					9100	73	6643				

Table A.11. Full-scale experimental results conducted on 22.05.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
22.05.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	30	NM	NM	NM	NM	-
			Bio-P_{inf}	281	740	248	40	22,3	ND	12,6	9,1	1,4
			Bio-P_{eff}	NM	NM	45	NM	NM	ND	71	20,0	ND
			Anoxic	NM	NM	50	NM	NM	1,2	NM	12,1	0,5
			Aerobic	NM	NM	NM	NM	NM	6,3	228	3,6	2,1
			Effluent	NM	55	48	10,1	1,4	7,24	4,5	3,8	3,2
			Recycle	NM	NM	41	NM	NM	6,1	NM	3,9	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
22.05.2006	Anaerobic	2,2	2500	1,8	2,72	3800	72	2736	8,2	NM	90	19,4
	Anoxic					3700	71	2627				
	Aerobic					3690	74	2730,6				
	Recycle					8500	75	6375				
	Excess					8500	75	6375				

Table A.12. Full-scale experimental results conducted on 03.06.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
03.06.2006	Sunny	I (PS Online)	Influent	NM	411	124	NM	NM	NM	NM	NM	-
			Bio-P_{inf}	148	370	130	37	NM	ND	9,6	5,4	1,5
			Bio-P_{eff}	NM	NM	35	NM	NM	ND	60	10,6	ND
			Anoxic	NM	NM	40	NM	NM	3,5	NM	4,9	0,5
			Aerobic	NM	NM	32	NM	NM	5,5	210	2,3	2,1
			Effluent	NM	45	35	5,4	0,7	4,1	2,8	2,4	3,2
			Recycle	NM	NM	40	NM	NM	3,2	NM	2,1	1,6

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. °C
03.06.2006	Anaerobic	2,3	2850	1,8	2,72	3600	70	2520	8,7	NM	70	22
	Anoxic					3550	69	2449,5				
	Aerobic					3450	69	2380,5				
	Recycle					8100	72	5832				
	Excess					8100	72	5832				

Table A.13. Full-scale experimental results conducted on 28.06.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
28.06.2006	Sunny	I (PS Online)	Influent	NM	380	NM	NM	26,8	NM	NM	NM	NM
			Bio-P_{inf}	176	480	155	32	25,3	NM	6,6	4,4	1,8
			Bio-P_{eff}	NM	NM	35	NM	NM	ND	55	11,6	ND
			Anoxic	NM	NM	40	NM	NM	1,5	NM	5,5	0,5
			Aerobic	NM	NM	36	NM	NM	3,8	232	0,9	2,1
			Effluent	NM	35	18	3,65	1,2	1,7	1,3	1,1	3,2
			Recycle	NM	NM	30	NM	NM	1,8	NM	1,0	1,5

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
28.06.2006	Anaerobic	2,1	3400	1,8	2,72	4000	70	2800	9,1	NM	98	26
	Anoxic					3880	69	2677,2				
	Aerobic					3800	67	2546				
	Recycle					8300	69	5727				
	Excess					8300	69	5727				

Table A.14. Full-scale experimental results conducted on 10.07.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
10.07.2006	Sunny	I (PS Online)	Influent	NM	380	NM	NM	26,8	NM	NM	NM	NM
			Bio-P_{inf}	92	310	120	28	25,3	NM	8,5	6,1	2,1
			Bio-P_{eff}	NM	NM	40	NM	NM	ND	62	9,1	ND
			Anoxic	NM	NM	40	NM	NM	3,9	NM	5,4	0,5
			Aerobic	NM	NM	36	NM	NM	3,5	140	4,7	2,1
			Effluent	NM	60	40	6,5	0,5	3,8	5,4	4,8	3,2
			Recycle	NM	NM	38	NM	NM	3,6	NM	4,5	1,6

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
10.07.2006	Anaerobic	1,6	3800	1,8	2,72	4600	74	3404	4,1	6%	115	25,8
	Anoxic					4500	74	3330				
	Aerobic					4500	74	3330				
	Recycle					8100	76	6156				
	Excess					8100	76	6156				

Table A.15. Full-scale experimental results conducted on 26.07.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
26.07.2006	Sunny	I (PS Online)	Influent	NM	450	NM	NM	26,8	NM	NM	NM	NM
			Bio-P_{inf}	130	420	125	32	27,4	NM	7,6	5,1	2,5
			Bio-P_{eff}	NM	NM	35	NM	64	ND	NM	7,6	ND
			Anoxic	NM	NM	38	NM	NM	3,4	NM	5,1	0,5
			Aerobic	NM	NM	42	NM	118	3,3	NM	4,0	2,1
			Effluent	NM	65	36	4,5	0,7	3,6	4,5	4,2	3,2
			Recycle	NM	NM	35	NM	NM	3,4	NM	4,1	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. °C
26.07.2006	Anaerobic	1,7	1300	1,8	2,72	3600	72	2592	4,7	NM	112	27
	Anoxic					3580	71	2541,8				
	Aerobic					3400	72	2448				
	Recycle					7100	72	5112				
	Excess					7100	72	5112				

Table A.16. Full-scale experimental results conducted on 02.08.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
02.08.2006	Sunny	I (PS Online)	Influent	NM	370	105	28	21,0	ND	7,6	NM	NM
			Bio-P_{inf}	98	340	105	26,5	NM	NM	7,9	4,9	2,1
			Bio-P_{eff}	NM	NM	40	NM	2,0	ND	72	8,1	ND
			Anoxic	NM	NM	35	NM	NM	2,1	NM	5,7	0,5
			Aerobic	NM	NM	36	NM	NM	3,1	123	4,7	2,1
			Effluent	NM	68	32	4,5	0,6	2,2	4,9	4,3	3,2
			Recycle	NM	NM	30	NM	NM	2,1	NM	4,6	1,7

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
02.08.2006	Anaerobic	1,6	4177	1,8	2,72	3880	72	2794	4,5	NM	100	27
	Anoxic					3750	72	2700				
	Aerobic					3680	72	2650				
	Recycle					7250	75	5438				
	Excess					7250	75	5438				

Table A.17. Full-scale experimental results conducted on 31.08.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
31.08.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	NM	NM	ND	NM	NM	NM
			Bio-P_{inf}	92	290	88	22	18,5	NM	8,4	6,2	2,5
			Bio-P_{eff}	NM	NM	36	NM	NM	ND	68	7,4	ND
			Anoxic	NM	NM	38	NM	NM	0	NM	5,8	0,2
			Aerobic	NM	NM	34	NM	NM	2,4	78	5,2	3,1
			Effluent	NM	35	28	4,8	0,7	3,1	5,9	5,6	3,6
			Recycle	NM	NM	27	NM	NM	2,7	NM	5,4	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
31.08.2006	Anaerobic	1,75	2100	1,8	2,72	3680	74	2723,2	2,9	6%	150	29
	Anoxic					3600	74	2664				
	Aerobic					3400	74	2516				
	Recycle					8480	75	6360				
	Excess					8480	75	6360				

Table A.18. Full-scale experimental results conducted on 08.09.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
08.09.2006	Sunny	I (PS Online)	Influent	NM	370	96	36	NM	ND	7,8	4,7	2,9
			Bio-P_{inf}	114	350	109	32	28	NM	7,7	4,8	2,1
			Bio-P_{eff}	NM	NM	38	NM	NM	ND	64	7,1	ND
			Anoxic	NM	NM	38	NM	NM	3,7	NM	5,1	0,3
			Aerobic	NM	NM	32	NM	NM	3,7	94	4,1	2,5
			Effluent	NM	44	32	5,1	0,6	4,1	4,6	4,2	3,6
			Recycle	NM	NM	36	NM	NM	3,1	NM	4,2	1,7

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
08.09.2006	Anaerobic	1,8	1940	1,8	2,72	3240	74	2397,6	4,0	6%	200	25
	Anoxic					3150	74	2331				
	Aerobic					3100	72	2232				
	Recycle					7100	75	5325				
	Excess					7100	75	5325				

Table A.19. Full-scale experimental results conducted on 27.09.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
27.09.2006	Sunny	I (PS Online)	Influent	NM	376	110	40	NM	ND	8,1	NM	2,7
			Bio-P_{inf}	115	350	109	32	24,6	NM	7,2	4,8	2,1
			Bio-P_{eff}	NM	NM	37	NM	NM	ND	63	7,1	ND
			Anoxic	NM	NM	35	NM	NM	3,7	NM	5,1	0,2
			Aerobic	NM	NM	36	NM	NM	3,7	94	4,1	2,4
			Effluent	NM	44	34	5,1	0,6	4,1	4,3	4,2	3,2
			Recycle	NM	NM	40	NM	NM	3,1	NM	4,2	1,7

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
27.09.2006	Anaerobic	1,8	1940	1,8	2,72	3240	74	2400	4,0	NM	180	23
	Anoxic					3150	74	2300				
	Aerobic					3100	72	2250				
	Recycle					7100	75	5350				
	Excess					7100	75	5350				

Table A.20. Full-scale experimental results conducted on 14.10.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
14.10.2006	Sunny	I (PS Online)	Influent	NM	NM	NM	36	NM	ND	7,8	4,7	2,9
			Bio-P_{inf}	118	340	103	22,1	16,7	NM	6,1	4,3	2,2
			Bio-P_{eff}	NM	NM	38	NM	NM	ND	53	7,5	ND
			Anoxic	NM	NM	30	NM	NM	1,5	NM	4,5	0,3
			Aerobic	NM	NM	32	NM	NM	2,3	130	2,4	2,5
			Effluent	NM	32	16	3,8	0,1	1,4	2,8	2,5	3,6
			Recycle	NM	NM	34	NM	NM	1,2	NM	2,2	1,7

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. °C
14.10.2006	Anaerobic	1,8	3745	1,8	2,72	3260	70	2282	6,0	NM	190	21,5
	Anoxic					3180	70	2226				
	Aerobic					3100	69	2139				
	Recycle					7710	71	5474				
	Excess					7710	71	5474				

Table A.24. Full-scale experimental results conducted on 30.11.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
30.11.2006	Sunny	III (PS out of order)	Influent	NM	810	NM	NM	NM	NM	NM	7,5	NM
			Bio-P_{inf}	310	770	296	40	33	NM	14,2	9,1	1,9
			Bio-P_{eff}	NM	NM	75	NM	NM	ND	62	24,0	ND
			Anoxic	NM	NM	63	NM	NM	0,6	NM	13,4	0,3
			Aerobic	NM	NM	65	NM	NM	2	214	2,1	2,5
			Effluent	NM	64	40	6,5	5,7	0,6	2,4	2,1	3,6
			Recycle	NM	NM	60	NM	NM	0,6	NM	2,1	1,6

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
30.11.2006	Anaerobic	1,8	4088	1,8	2,72	3040	74	2249,6	9,9	NM	160	19,4
	Anoxic					2930	74	2168,2				
	Aerobic					2900	74	2146				
	Recycle					8470	75	6352,5				
	Excess					8470	75	6352,5				

Table A.21. Full-scale experimental results conducted on 08.11.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
08.11.2006	Sunny	I (PS Online)	Influent	NM	710	NM	48	36,0	NM	12,4	6,4	NM
			Bio-P_{inf}	205	585	190	40,1	16,7	NM	11,3	5,6	2,5
			Bio-P_{eff}	NM	NM	92	NM	NM	ND	60	9,2	ND
			Anoxic	NM	NM	84	NM	NM	4,1	NM	5,9	0,3
			Aerobic	NM	NM	68	NM	NM	4,3	165	4,0	2,5
			Effluent	NM	90	60	6,5	2,1	3,8	4,5	4,1	3,6
			Recycle	NM	NM	65	NM	NM	3,1	NM	4,1	1,8

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. °C
08.11.2006	Anaerobic	1,8	2955	1,8	2,72	3680	80	2944	5,6	NM	200	19,7
	Anoxic					3600	81	2916				
	Aerobic					3580	80	2864				
	Recycle					6280	82	5149,6				
	Excess					6280	82	5149,6				

Table A.22. Full-scale experimental results conducted on 15.11.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
15.11.2006	Sunny	I (PS Online)	Influent	NM	740	NM	32,1	21,6	NM	10,2	5,4	NM
			Bio-P_{inf}	224	600	220	28,1	16,7	NM	11,5	6,1	2,2
			Bio-P_{eff}	NM	NM	96	NM	NM	ND	58	10,6	ND
			Anoxic	NM	NM	NM	NM	NM	2,4	NM	7,0	0,3
			Aerobic	NM	NM	64	NM	NM	6,5	110	4,1	2,5
			Effluent	NM	84	62	5,5	0,1	2,2	4,4	3,9	3,6
			Recycle	NM	NM	72	NM	NM	2,1	NM	4,1	1,6

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
15.11.2006	Anaerobic	1,75	2615	1,8	2,72	3280	64	2099,2	5,2	NM	163	19,8
	Anoxic					3260	64	2086,4				
	Aerobic					3210	64	2054,4				
	Recycle					6540	65	4251				
	Excess					6540	65	4251				

Table A.23. Full-scale experimental results conducted on 24.11.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
24.11.2006	Sunny	III (PS out of order)	Influent	NM	780	290	40	NM	NM	10,4	7,5	NM
			Bio-P_{inf}	286	740	310	37	33	NM	12,9	7,5	1,9
			Bio-P_{eff}	NM	NM	84	NM	NM	ND	62	19,9	ND
			Anoxic	NM	NM	53	NM	NM	3,1	NM	10,4	0,3
			Aerobic	NM	NM	64	NM	NM	2,7	250	1,0	2,5
			Effluent	NM	76	48	4,2	0,86	3,1	1,5	1,0	3,6
			Recycle	NM	NM	56	NM	NM	3,2	NM	1,3	1,5

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
24.11.2006	Anaerobic	1,75	4088	1,8	2,72	3300	74	2442	10,5	NM	148	19,4
	Anoxic					3240	74	2397,6				
	Aerobic					3200	74	2368				
	Recycle					8280	75	6210				
	Excess					8280	75	6210				

Table A.24. Full-scale experimental results conducted on 30.11.2006

Date	Weather Condition	Line	Sampling point	BOD mg/L	COD mg/L	sCOD mg/L	TN mg/L	NH_4 mg/L	NO_3 mg/L	TP mg/L	PO_4 mg/L	DO mg/L
30.11.2006	Sunny	III (PS out of order)	Influent	NM	810	NM	NM	NM	NM	NM	7,5	NM
			Bio-P_{inf}	310	770	296	40	33	NM	14,2	9,1	1,9
			Bio-P_{eff}	NM	NM	75	NM	NM	ND	62	24,0	ND
			Anoxic	NM	NM	63	NM	NM	0,6	NM	13,4	0,3
			Aerobic	NM	NM	65	NM	NM	2	214	2,1	2,5
			Effluent	NM	64	40	6,5	5,7	0,6	2,4	2,1	3,6
			Recycle	NM	NM	60	NM	NM	0,6	NM	2,1	1,6

Date	Sampling point	Q_{inf} m^3/s	Q_{es} m^3/s	Q_{rxl} m^3/s	Q_{int} m^3/s	MLSS mg/L	LOI %	MLVSS mg/L	Mixed Sludge P Content %	Mixed Sludge N Content %	SVI ml/g	TEMP. ^oC
30.11.2006	Anaerobic	1,8	4088	1,8	2,72	3040	74	2249,6	9,9	NM	160	19,4
	Anoxic					2930	74	2168,2				
	Aerobic					2900	74	2146				
	Recycle					8470	75	6352,5				
	Excess					8470	75	6352,5				

APPENDIX- B

Raw Data for Full-Scale (Second Part) & Batch-Scale Investigations

Table B.1. Influent characterization results conducted between 01.11 – 22.11 / 2006

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$TP_{inf-III}$	TP_{inf-I}
	Q_I	Q_{III}	^oC	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
01.11.2006	2,22	2,18	16,60	7,71	7,69	563	-	-	-	-	-	-	-
02.11.2006	2,74	2,89	20,40	7,63	7,65	403	-	-	-	-	-	14,2	-
03.11.2006	2,93	2,93	18,50	7,35	7,32	468	-	-	-	-	-	7,1	-
04.11.2006	3,00	3,00	18,70	7,24	7,22	304	-	-	-	-	-	7,3	-
05.11.2006	2,52	2,60	18,10	7,45	7,42	550	390	166	165	65	62	7,8	7,0
06.11.2006	2,52	2,54	18,00	7,48	7,46	497	360	140	132	58	57	12,0	10,8
07.11.2006	2,46	2,45	19,90	7,35	7,36	667	366	205	210	80	79	13,7	12,3
08.11.2006	2,36	2,38	19,70	7,20	7,25	655	450	198	190	79	76	13,3	12,0
09.11.2006	2,40	2,33	20,00	7,42	7,40	742	521	220	218	82	82	10,8	9,7
10.11.2006	2,40	2,42	20,10	7,42	7,48	651	-	-	-	-	-	9,7	-
11.11.2006	2,31	2,30	20,00	7,42	7,42	372	-	-	-	-	-	7,2	-
12.11.2006	2,45	2,52	19,50	7,40	7,40	350	-	-	-	-	-	7,6	-
13.11.2006	2,52	2,38	19,80	7,36	7,34	285	-	-	-	-	-	7,2	-
14.11.2006	2,43	2,30	19,30	7,17	7,18	755	528	227	225	82	76	8,1	7,3
15.11.2006	2,29	2,33	19,90	7,02	7,04	628	420	188	188	75	73	10,1	9,1
16.11.2006	2,24	2,28	20,90	7,21	7,28	753	520	226	224	90	85	13,4	12,1
17.11.2006	2,25	2,31	19,00	7,21	7,24	552	380	166	160	64	62	12,2	10,9
18.11.2006	2,34	2,27	19,20	7,26	7,24	602	420	181	176	70	61	12,9	11,6
19.11.2006	2,33	2,28	19,32	7,28	7,26	550	-	-	-	-	-	10,5	-
20.11.2006	2,27	2,35	19,30	7,21	7,23	446	-	-	-	-	-	10,2	-
21.11.2006	2,23	2,36	20,00	7,35	7,36	476	-	-	-	-	-	13,2	-
22.11.2006	2,22	2,35	19,30	7,35	7,31	709	-	-	-	-	-	14,0	-

Table B.2. Influent characterization results conducted between 23.11 – 14.12 / 2006

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$Tp_{inf-III}$	Tp_{inf-I}
	Q_I	Q_{III}	°C	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
23.11.2006	2,58	2,60	18,10	7,02	6,98	764	535	230	226	92	90	14,1	12,7
24.11.2006	2,30	2,33	19,40	7,41	7,41	728	510	230	233	92	93	10,4	9,4
25.11.2006	2,40	2,45	20,00	7,36	7,31	526	368	160	165	70	70	11,2	10,1
26.11.2006	2,35	2,36	20,00	7,38	7,34	650	-	-	-	-	-	11,3	-
27.11.2006	2,30	2,34	19,90	7,35	7,42	750	-	-	-	-	-	11,0	-
28.11.2006	2,24	2,22	20,20	7,31	7,30	454	-	-	-	-	-	11,7	-
29.11.2006	2,25	2,21	19,10	7,55	7,66	980	688	300	293	110	106	5,5	4,9
30.11.2006	2,22	2,21	19,10	7,58	7,55	492	340	140	136	62	64	13,1	11,8
01.12.2006	2,22	2,15	18,50	7,31	7,31	686	478	205	201	78	82	10,8	9,7
02.12.2006	2,19	2,19	18,40	7,44	7,46	492	340	146	143	61	60	10,7	9,6
03.12.2006	2,22	2,23	18,40	7,56	7,52	681	-	-	-	-	-	10,6	-
04.12.2006	2,20	2,24	18,20	7,49	7,52	750	-	-	-	-	-	13,6	-
05.12.2006	2,16	2,16	17,80	7,43	7,44	521	-	-	-	-	-	10,4	-
06.12.2006	2,18	2,14	18,60	7,52	7,56	587	410	180	176	72	67	10,2	9,1
07.12.2006	2,20	2,20	17,80	7,25	7,28	513	359	155	154	62	60	10,2	9,1
08.12.2006	2,20	2,17	18,60	7,30	7,26	495	347	148	148	58	55	10,1	9,1
09.12.2006	2,28	2,19	18,70	7,35	7,48	526	368	157	166	51	52	10,3	9,3
10.12.2006	2,24	2,25	17,50	7,45	7,44	556	-	-	-	-	-	10,5	-
11.12.2006	2,22	2,25	17,30	7,41	7,41	577	-	-	-	-	-	12,5	-
12.12.2006	2,24	2,20	17,80	7,58	7,62	555	-	-	-	-	-	10,9	-
13.12.2006	2,21	2,20	19,00	7,70	7,68	653	457	192	192	77	77	11,0	9,9
14.12.2006	2,07	2,11	19,00	7,56	7,58	422	295	127	127	55	56	9,8	8,8

Table B.3. Influent characterization results conducted between 15.12.2006 – 05.01. 2007

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$Tp_{inf-III}$	Tp_{inf-I}
	Q_I	Q_{III}	^oC	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
15.12.2006	2,10	2,12	18,40	7,68	7,64	542	379	163	163	66	62	12,0	10,8
16.12.2006	2,17	2,19	17,40	7,70	7,72	613	429	184	184	74	73	13,3	11,9
17.12.2006	2,18	2,19	18,00	7,76	7,71	645	452	198	198	76	74	13,6	12,2
18.12.2006	2,21	2,24	18,00	7,54	7,51	804	-	-	-	-	-	17,8	-
19.12.2006	2,21	2,20	18,00	7,53	7,56	732	-	-	-	-	-	13,2	-
20.12.2006	2,32	2,31	18,20	7,67	7,68	734	-	-	-	-	-	11,9	-
21.12.2006	2,23	2,23	18,30	7,69	7,69	1250	-	-	-	-	-	15,3	-
22.12.2006	2,18	2,10	17,60	7,62	7,64	392	274	120	118	50	48	10,1	9,1
23.12.2006	2,20	2,17	17,00	7,60	7,60	511	358	155	156	62	69	12,1	10,9
24.12.2006	2,22	2,19	16,80	7,50	7,50	586	-	-	-	-	-	11,6	-
25.12.2006	2,22	2,18	16,60	7,71	7,72	563	394	170	163	68	63	11,7	10,5
26.12.2006	2,14	2,09	19,70	7,58	7,56	869	608	262	268	98	84	12,4	11,1
27.12.2006	2,07	2,10	16,00	7,56	7,55	664	465	200	206	80	78	10,2	9,2
28.12.2006	2,08	2,04	15,80	7,87	7,83	687	-	-	-	-	-	10,7	-
29.12.2006	2,20	2,20	15,70	7,51	7,53	512	-	-	-	-	-	12,8	-
30.12.2006	2,19	2,22	16,20	7,67	7,65	584	-	-	-	-	-	13,1	-
31.12.2006	2,20	2,21	16,50	7,56	7,55	620	-	-	-	-	-	10,5	-
01.01.2007	2,28	2,26	16,54	7,65	7,61	606	424	180	190	72	76	10,8	9,7
02.01.2007	2,26	2,25	16,52	7,46	7,48	652	456	195	194	78	78	10,7	9,6
03.01.2007	2,27	2,28	15,80	7,56	7,56	586	410	175	173	70	69	10,2	9,2
04.01.2007	2,47	2,48	15,20	7,65	7,62	1159	-	-	-	-	-	13,4	-
05.01.2007	2,21	2,19	16,00	7,55	7,51	563	-	-	-	-	-	12,7	-

Table B.4. Influent characterization results conducted between 06.01-27.01/ 2007

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$Tp_{inf-III}$	Tp_{inf-I}
	Q_I	Q_{III}	^oC	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
06.01.2007	2,19	2,22	15,50	7,73	7,74	606	424	180	164	72	66	13,6	12,2
07.01.2007	2,20	2,20	18,50	7,45	7,43	584	409	175	178	70	71	10,6	9,5
08.01.2007	2,37	2,30	20,60	7,69	7,67	421	295	126	124	51	50	7,0	6,3
09.01.2007	2,45	2,40	19,50	7,62	7,61	966	676	290	285	105	105	14,3	12,9
10.01.2007	2,16	2,19	17,60	7,74	7,78	739	517	222	221	88	88	9,1	8,2
11.01.2007	2,20	2,19	17,80	7,76	7,74	634	-	-	-	-	-	9,6	-
12.01.2007	2,22	2,20	16,60	7,63	7,58	564	-	-	-	-	-	10,7	-
13.01.2007	2,25	2,26	16,50	7,64	7,56	692	-	-	-	-	-	10,8	-
14.01.2007	2,21	2,23	16,50	7,68	7,72	680	-	-	-	-	-	10,6	-
15.01.2007	2,15	2,16	17,20	7,69	7,68	397	290	119	115	45	48	6,0	5,4
16.01.2007	2,09	2,09	17,20	7,61	7,56	742	510	223	225	80	72	12,2	11,0
17.01.2007	1,99	2,07	16,60	7,72	7,76	822	570	250	248	98	96	11,6	10,5
18.01.2007	2,13	2,17	18,00	7,55	7,51	716	500	213	212	86	83	10,8	9,7
19.01.2007	2,14	2,12	17,30	7,62	7,68	707	-	-	-	-	-	11,1	-
20.01.2007	2,14	2,17	17,40	7,61	7,63	783	-	-	-	-	-	12,1	-
21.01.2007	2,14	2,16	17,50	7,65	7,66	801	-	-	-	-	-	12,3	-
22.01.2007	2,19	2,16	17,60	7,79	7,81	855	-	-	-	-	-	11,9	-
23.01.2007	2,06	2,09	17,90	7,49	7,52	664	465	201	196	80	75	10,9	9,8
24.01.2007	2,08	2,14	17,90	7,63	7,66	620	434	198	210	79	84	11,0	9,9
25.01.2007	2,08	2,17	17,80	7,64	7,63	698	-	-	-	-	-	10,3	-
26.01.2007	2,09	2,15	18,10	7,62	7,58	715	-	-	-	-	-	7,3	-
27.01.2007	1,98	1,95	17,60	7,64	7,68	701	491	211	206	84	82	7,9	7,1

Table B.5. Influent characterization results conducted between 28.01-18.02/ 2007

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$Tp_{inf-III}$	Tp_{inf-I}
	Q_I	Q_{III}	^oC	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
28.01.2007	2,15	2,15	16,80	7,62	7,61	756	529	225	224	90	90	8,2	7,4
29.01.2007	2,15	2,15	15,90	7,70	7,70	898	-	-	-	-	-	11,1	-
30.01.2007	2,25	2,25	15,70	7,67	7,66	339	-	-	-	-	-	7,8	-
31.01.2007	2,10	2,11	16,40	7,68	7,66	379	-	-	-	-	-	7,4	-
01.02.2007	2,13	2,14	15,80	7,77	7,78	668	-	-	-	-	-	11,3	-
02.02.2007	2,53	2,59	15,50	7,68	7,64	942	660	282	280	113	112	6,3	6,0
03.02.2007	2,06	2,06	15,30	7,66	7,63	813	550	244	236	98	94	9,7	9,0
04.02.2007	2,06	2,06	15,20	7,65	7,65	623	430	190	186	76	74	9,8	9,1
05.02.2007	2,06	2,06	14,60	7,63	7,64	585	410	175	173	70	69	9,3	8,6
06.02.2007	2,02	2,06	15,50	7,71	7,74	609	-	-	-	-	-	9,9	-
07.02.2007	2,06	2,07	14,90	7,69	7,68	711	-	-	-	-	-	10,7	-
08.02.2007	2,11	2,10	17,00	7,72	7,73	601	-	-	-	-	-	9,5	-
09.02.2007	2,04	2,11	16,30	7,70	7,69	753	520	226	224	86	81	10,4	9,2
10.02.2007	2,17	2,19	16,60	7,73	7,71	786	550	236	237	88	92	10,6	9,3
11.02.2007	2,18	2,19	16,50	7,56	7,58	660	-	-	-	-	-	10,1	-
12.02.2007	2,17	2,16	18,50	7,82	7,87	538	-	-	-	-	-	12,5	-
13.02.2007	2,13	2,20	17,80	7,31	7,36	611	427	180	176	76	71	10,9	10,1
14.02.2007	2,57	2,55	16,70	7,60	7,62	746	522	220	218	84	78	11,6	11,1
15.02.2007	2,30	2,34	16,50	7,61	7,59	682	-	-	-	-	-	10,4	-
16.02.2007	2,20	2,23	15,90	7,63	7,65	557	-	-	-	-	-	9,9	-
17.02.2007	2,20	2,24	17,00	7,56	7,58	564	-	-	-	-	-	10,1	-
18.02.2007	2,20	2,24	16,50	7,65	7,62	640	-	-	-	-	-	10,6	-

Table B.6. Influent characterization results conducted between 19.02-12.03/ 2007

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$Tp_{inf-III}$	Tp_{inf-I}
	Q_I	Q_{III}	^oC	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
19.02.2007	2,15	2,16	16,30	7,56	7,55	661	-	-	-	-	-	11,8	-
20.02.2007	2,06	2,10	18,30	7,52	7,54	437	300	130	128	56	48	7,3	6,8
21.02.2007	2,11	2,10	17,80	7,61	7,63	632	440	190	185	76	73	10,2	9,6
22.02.2007	2,08	2,08	17,10	7,60	7,58	642	450	193	193	75	75	10,4	9,6
23.02.2007	2,11	2,13	13,80	7,64	7,66	622	435	190	190	76	82	10,2	10,1
24.02.2007	2,12	2,14	16,40	7,56	7,58	578	410	175	173	70	69	10,3	9,6
25.02.2007	2,15	2,16	17,00	7,58	7,59	550	-	-	-	-	-	8,6	-
26.02.2007	2,08	2,09	17,00	7,61	7,62	496	-	-	-	-	-	7,6	-
27.02.2007	2,14	2,19	17,00	7,64	7,64	501	-	-	-	-	-	7,8	-
28.02.2007	2,30	2,38	16,50	7,48	7,50	863	-	-	-	-	-	8,7	-
01.03.2007	2,08	2,12	17,40	7,60	7,61	527	-	-	-	-	-	8,0	-
02.03.2007	2,07	2,13	17,70	7,41	7,38	487	310	150	148	60	59	7,8	7,1
03.03.2007	2,11	2,13	17,70	7,42	7,41	516	385	155	154	62	53	7,6	6,6
04.03.2007	2,18	2,16	17,60	7,56	7,59	565	400	175	174	70	74	7,9	7,1
05.03.2007	2,09	2,11	18,80	7,63	7,58	525	370	160	158	64	64	11,8	10,2
06.03.2007	2,01	2,05	18,80	7,52	7,54	687	480	205	201	82	80	11,2	9,8
07.03.2007	2,03	2,07	18,80	7,60	7,58	681	-	-	-	-	-	10,8	-
08.03.2007	2,03	2,03	19,20	7,69	7,66	630	-	-	-	-	-	10,0	-
09.03.2007	2,05	2,07	18,90	7,39	7,42	539	-	-	-	-	-	7,9	-
10.03.2007	2,10	2,12	18,30	7,46	7,44	585	-	-	-	-	-	8,1	-
11.03.2007	2,10	2,12	18,50	7,56	7,61	680	476	205	203	82	81	8,3	7,5
12.03.2007	2,10	2,12	19,20	7,43	7,46	782	550	235	241	94	96	14,5	12,2

Table B.7. Influent characterization results conducted between 13.03-31.03/ 2007

Date	Flow m^3/s		Temp.	pH_{inf-I}	$pH_{inf-III}$	$COD_{inf-IIII}$	COD_{inf-I}	$rbsCOD_{inf-III}$	$rbsCOD_{inf-I}$	$VFAs_{inf-III}$	$VFAs_{inf-I}$	$Tp_{inf-III}$	Tp_{inf-I}
	Q_I	Q_{III}	^oC	-	-	mg/L	mg/L	mg/L	mg/L	mgHAC/L	mgHAC/L	mg/L	mg/L
13.03.2007	1,98	2,00	13,80	7,75	7,73	731	512	220	218	88	87	10,9	10,1
14.03.2007	1,97	1,98	16,60	7,73	7,74	658	460	198	191	79	76	11,6	10,2
15.03.2007	2,02	1,92	17,30	7,50	7,52	604	430	180	176	72	70	9,3	8,8
16.03.2007	2,08	1,94	18,00	7,55	7,55	715	500	216	224	84	84	10,5	9,6
17.03.2007	2,11	1,99	18,30	7,69	7,71	772	-	-	-	-	-	11,1	-
18.03.2007	2,16	2,11	18,60	7,71	7,71	772	-	-	-	-	-	12,1	-
19.03.2007	2,07	2,06	18,70	7,70	7,66	772	-	-	-	-	-	14,2	-
20.03.2007	1,99	1,97	19,20	7,67	7,66	772	-	-	-	-	-	10,9	-
21.03.2007	2,05	2,02	19,10	7,59	7,61	772	-	-	-	-	-	11,9	-
22.03.2007	2,13	2,12	18,00	7,60	7,64	772	-	-	-	-	-	10,1	-
23.03.2007	2,45	2,44	16,30	7,68	7,65	772	562	232	284	102	98	11,8	9,6
24.03.2007	2,64	2,68	16,50	7,65	7,64	772	540	232	262	100	106	7,1	6,5
25.03.2007	2,20	2,20	17,50	7,66	7,63	772	533	230	228	86	84	9,5	8,6
26.03.2007	2,16	2,17	18,50	7,67	7,64	772	540	205	203	78	71	11,7	9,6
27.03.2007	2,12	2,07	18,50	7,69	7,71	772	-	-	-	-	-	10,0	-
28.03.2007	2,11	2,02	18,10	7,58	7,59	772	-	-	-	-	-	19,3	-
29.03.2007	2,32	2,30	18,00	7,55	7,53	772	-	-	-	-	-	11,3	-
30.03.2007	2,06	2,08	18,00	7,59	7,61	772	480	232	201	78	72	11,4	10,1
31.03.2007	2,12	2,10	17,90	7,56	7,56	772	460	198	193	80	88	11,1	10,2

Table B.8. Process characterization results conducted between 01.11-20.11/ 2006

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
01.11.2006	2870	2191	76	209	9	2680	1785	67	194	6
02.11.2006	3060	2325	76	210	9	2350	1572	67	221	6
03.11.2006	2490	1848	74	199	8	2410	1641	68	183	6
04.11.2006	2820	2138	76	220	7	2450	1567	64	170	6
05.11.2006	2810	2108	75	230	7	2560	1741	68	160	6
06.11.2006	2830	2123	75	240	7	2720	1831	67	162	6
07.11.2006	2810	2072	74	204	7	2850	1911	67	196	6
08.11.2006	2750	1999	73	156	7	2740	1990	73	190	6
09.11.2006	3600	2961	82	182	8	3050	2024	66	157	7
10.11.2006	2860	2177	76	161	9	3010	2011	67	173	9
11.11.2006	3480	2553	73	156	11	2790	1964	70	172	9
12.11.2006	3330	2513	75	165	12	3100	2232	72	160	9
13.11.2006	3440	2477	72	189	12	3440	2464	72	151	9
14.11.2006	3590	2527	70	175	12	2870	2049	71	167	9
15.11.2006	3200	2467	77	199	12	2930	1819	62	164	10
16.11.2006	3210	2064	64	171	13	3190	2206	69	176	10
17.11.2006	3740	2738	73	167	12	2910	1997	69	165	10
18.11.2006	3560	2599	73	155	11	3100	2170	70	170	10
19.11.2006	3550	2592	73	160	11	2980	2116	71	175	10
20.11.2006	3280	2394	73	155	10	2860	2059	72	176	10

Table B.9. Process characterization results conducted between 21.11-10.12/ 2006

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
21.11.2006	3200	2345	73	155	8	2890	2115	73	180	10
22.11.2006	3350	2507	75	152	8	3180	2373	75	151	9
23.11.2006	3690	2733	74	161	8	2870	2014	70	167	9
24.11.2006	3230	2406	74	153	8	3240	2316	71	148	9
25.11.2006	3400	2538	75	171	9	3370	2529	75	142	9
26.11.2006	3300	2475	75	170	9	3460	2526	73	140	10
27.11.2006	3200	2368	74	170	9	3200	2299	72	160	12
28.11.2006	3170	2330	73	165	11	3020	2004	66	170	12
29.11.2006	3390	2502	74	173	11	2690	1874	70	178	11
30.11.2006	3240	2402	74	149	11	2930	2134	73	164	11
01.12.2006	3480	2626	75	170	11	2840	2128	75	162	11
02.12.2006	3300	2475	75	178	11	2770	2078	75	173	10
03.12.2006	3200	2432	76	160	11	2780	2085	75	165	10
04.12.2006	3250	2568	79	157	11	2780	2091	75	158	7
05.12.2006	3560	2796	79	178	10	3100	2310	75	155	7
06.12.2006	3590	2748	77	171	10	3440	2619	76	169	7
07.12.2006	3280	2545	78	167	11	2820	2104	75	156	7
08.12.2006	3360	2592	77	153	10	2750	1882	68	160	7
09.12.2006	3280	2526	77	155	10	2680	1876	70	155	7
10.12.2006	3260	2478	76	153	10	2710	2033	75	150	7

Table B.10. Process characterization results conducted between 11.12-30.12/ 2006

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
11.12.2006	3260	2510	77	140	10	2640	1954	74	150	8
12.12.2006	3260	2537	78	138	10	2620	1963	75	153	8
13.12.2006	3180	2418	76	156	9	2730	1974	72	147	8
14.12.2006	3080	2324	75	166	9	2680	2034	76	164	8
15.12.2006	3140	2405	77	165	9	2970	2184	74	146	9
16.12.2006	3240	2457	76	162	9	2820	1998	71	142	9
17.12.2006	3380	2535	75	160	9	2810	2051	73	145	9
18.12.2006	3450	2622	76	158	9	2810	2055	73	149	9
19.12.2006	3550	2623	74	144	9	3000	2238	75	127	9
20.12.2006	3610	2779	77	160	10	3010	2301	76	133	9
21.12.2006	3740	2890	77	146	10	3590	2348	65	111	9
22.12.2006	3570	2460	69	160	9	3730	2717	73	118	9
23.12.2006	3500	2610	75	171	9	3400	2385	70	141	9
24.12.2006	3600	2664	74	165	10	3480	2471	71	148	9
25.12.2006	3780	2835	75	160	10	3420	2479	72	152	9
26.12.2006	3740	2806	75	173	10	2690	1902	71	149	9
27.12.2006	3710	2752	74	156	10	2520	1822	72	143	9
28.12.2006	3340	2493	75	179	9	2620	1918	73	137	8
29.12.2006	3130	2361	75	158	9	2560	1862	73	125	8
30.12.2006	3040	2248	74	146	8	2650	1961	74	136	7

Table B.11. Process characterization results conducted between 31.12.2006 - 19.01.2007

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
31.12.2006	2950	2213	75	155	8	2800	2100	75	125	9
01.01.2007	2780	2085	75	150	8	3100	2294	74	125	9
02.01.2007	2640	1980	75	160	8	3280	2427	74	110	9
03.01.2007	2550	1913	75	160	8	3400	2516	74	110	11
04.01.2007	2450	1860	76	163	8	3330	2452	74	108	13
05.01.2007	2710	2048	76	192	8	3510	2570	73	114	10
06.01.2007	2980	2262	76	174	8	3630	2792	77	121	9
07.01.2007	2950	2242	76	175	9	3780	2911	77	122	7
08.01.2007	2930	2270	77	168	10	3850	2869	75	122	8
09.01.2007	4340	3392	78	165	12	4570	3402	74	122	9
10.01.2007	2740	2273	83	161	10	3290	2474	75	122	9
11.01.2007	3190	2452	77	163	9	3290	2477	75	134	9
12.01.2007	3200	2458	77	163	10	3310	2463	74	121	9
13.01.2007	3090	2424	78	168	12	2940	2219	75	136	9
14.01.2007	3050	2379	78	170	12	3100	2325	75	140	9
15.01.2007	3040	2364	78	170	12	3130	2359	75	140	9
16.01.2007	3670	2795	76	175	12	2850	2160	76	140	8
17.01.2007	3170	2428	77	177	11	3010	2269	75	133	8
18.01.2007	3200	2753	86	163	13	3340	2934	88	110	7
19.01.2007	3120	2393	77	167	10	3440	2657	77	105	7

Table B.12. Process characterization results conducted between 20.11 – 08.02 / 2007

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
20.01.2007	3080	2388	78	195	10	3230	2513	78	111	7
21.01.2007	3100	2418	78	190	10	3280	2558	78	110	7
22.01.2007	3110	2383	77	180	9	3380	2602	77	95	7
23.01.2007	3300	2385	72	165	13	3740	2774	74	96	7
24.01.2007	3510	2703	77	156	12	3640	2766	76	90	7
25.01.2007	3120	2425	78	167	13	3550	2642	74	90	8
26.01.2007	2920	2219	76	178	11	3570	2642	74	90	8
27.01.2007	3230	2378	74	161	10	3780	2807	74	71	8
28.01.2007	3250	2373	73	160	11	4150	2905	70	73	9
29.01.2007	3330	2422	73	155	12	4630	3159	68	72	9
30.01.2007	3120	2273	73	154	10	4890	3305	68	72	9
31.01.2007	3530	2641	75	147	12	5000	3476	70	74	10
01.02.2007	3170	2322	73	177	10	4590	3111	68	71	10
02.02.2007	3370	2446	73	154	12	4260	2934	69	72	10
03.02.2007	3420	2525	74	164	14	4350	3147	72	72	10
04.02.2007	3600	2664	74	165	12	4410	3087	70	72	9
05.02.2007	3740	2783	74	160	11	4490	3057	68	75	8
06.02.2007	4130	2994	72	155	11	4260	2923	69	75	7
07.02.2007	3010	2203	73	199	9	4130	2811	68	77	7
08.02.2007	3270	2496	76	183	12	4760	3367	71	70	7

Table B.13. Process characterization results conducted between 09.02 – 01.03 / 2007

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
09.02.2007	3450	2564	74	162	11	4410	3438	78	73	7
10.02.2007	3270	2386	73	183	9	4550	3209	71	79	8
11.02.2007	3330	2464	74	160	9	4300	3225	75	80	8
12.02.2007	3260	2415	74	150	8	4380	3121	71	75	8
13.02.2007	3160	2361	75	139	9	4440	3164	71	72	9
14.02.2007	3240	2446	75	148	10	5340	3831	72	70	10
15.02.2007	2820	2085	74	142	14	4690	3259	69	72	10
16.02.2007	3470	2486	72	138	9	4840	3269	68	71	11
17.02.2007	3470	2458	71	161	9	4840	3273	68	74	11
18.02.2007	3380	2366	70	155	9	4740	3223	68	75	11
19.02.2007	3330	2335	70	155	9	4680	3188	68	74	11
20.02.2007	3500	2465	70	149	8	5200	3879	75	85	12
21.02.2007	3540	2679	76	140	11	5030	3590	71	76	12
22.02.2007	3430	2564	75	140	10	4960	3488	70	81	11
23.02.2007	4130	2948	71	130	11	5290	3625	69	91	11
24.02.2007	3750	2756	73	140	14	4870	3406	70	107	11
25.02.2007	3200	2400	75	140	14	4480	3226	72	110	11
26.02.2007	2990	2257	75	145	13	4380	3262	74	105	11
27.02.2007	3300	2417	73	145	14	4510	3203	71	106	10
28.02.2007	3580	2590	72	123	12	4810	3276	68	100	10
01.03.2007	3780	2759	73	130	11	5070	3643	72	100	10

Table B.14. Process characterization results conducted between 02.03 – 21.03 / 2007

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
02.03.2007	4060	2984	73	138	9	5080	3665	72	95	10
03.03.2007	3510	2594	74	131	10	4760	3441	72	91	10
04.03.2007	3380	2529	75	130	10	4810	3463	72	97	10
05.03.2007	3450	2588	75	140	11	5350	3818	71	100	10
06.03.2007	3690	2806	76	150	11	5030	3486	69	100	11
07.03.2007	3490	2509	72	150	11	5080	3576	70	111	11
08.03.2007	3440	2515	73	140	12	4870	3569	73	102	11
09.03.2007	3800	2924	77	158	14	4550	3372	74	107	12
10.03.2007	3340	2568	77	155	12	4220	3009	71	110	13
11.03.2007	3650	2735	75	153	11	3860	2779	72	110	12
12.03.2007	3640	2730	75	163	12	3720	2740	74	110	12
13.03.2007	3660	2793	76	170	13	3760	2768	74	105	12
14.03.2007	3310	2469	75	193	10	3760	2735	73	106	11
15.03.2007	3310	2483	75	181	10	3760	2727	73	117	11
16.03.2007	3310	2428	73	180	12	3760	2751	73	128	10
17.03.2007	3310	2382	72	175	13	4100	2973	73	120	10
18.03.2007	3500	2567	73	171	15	4210	3073	73	117	10
19.03.2007	3600	2628	73	154	14	4280	3132	73	110	12
20.03.2007	3900	2807	72	150	13	4390	3232	74	110	12
21.03.2007	4160	3110	75	144	14	4210	3081	73	109	13

Table B.15. Process characterization results conducted between 22.03 – 31.03 / 2007

Date	I Line					III Line				
	$MLSS_I$	$MLVSS_I$	LOI_I	SVI_I	SRT_I	$MLSS_{III}$	$MLVSS_{III}$	LOI_{III}	SVI_{III}	SRT_{III}
	mg/L	mg/L	%	ml/g	days	mg/L	mg/L	%	ml/g	days
22.03.2007	3480	2505	72	149	14	4660	3410	73	114	14
23.03.2007	3490	2512	72	149	16	4470	3361	75	112	15
24.03.2007	3270	2512	77	159	15	4610	3372	73	107	17
25.03.2007	3160	2373	75	165	15	4320	3154	73	104	17
26.03.2007	3230	2390	74	160	15	4120	3023	73	110	18
27.03.2007	3370	2468	73	150	16	4310	3164	73	120	18
28.03.2007	3650	2746	75	153	17	4220	3051	72	121	18
29.03.2007	3710	2712	73	173	17	4480	3281	73	118	18
30.03.2007	3580	2637	74	168	15	4370	3048	70	125	18
31.03.2007	3690	2702	73	168	13	3990	2793	70	114	16

Table B.16. Line I effluent characterization results conducted between 01.11 – 15.12 / 2006

Date	COD_{eff-I}	NO_3-N_{eff-I}	DO_{inf-I}	$DO_{recyle-I}$	Total EA_{inf-I}	PO_4-P_{eff-I}
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
01.11.2006	110	2,10	1,92	1,80	1,6	4,22
02.11.2006	105	2,10	1,96	1,70	1,5	2,58
03.11.2006	100	1,80	1,86	1,80	1,3	4,65
04.11.2006	110	1,60	2,21	2,10	1,4	5,29
05.11.2006	100	2,10	2,23	2,10	1,6	4,85
06.11.2006	63	1,60	2,16	1,75	1,4	4,20
07.11.2006	42	0,35	1,68	1,86	0,8	4,12
08.11.2006	95	2,48	1,76	1,56	1,7	3,29
09.11.2006	30	2,10	1,96	1,85	1,6	3,18
10.11.2006	35	2,20	1,94	1,86	1,6	4,16
11.11.2006	39	2,10	1,86	1,92	1,6	3,24
12.11.2006	100	1,80	2,40	2,30	1,6	3,86
13.11.2006	120	2,10	2,20	2,10	1,6	4,25
14.11.2006	35	2,10	1,88	2,20	1,6	3,32
15.11.2006	56	2,50	1,98	2,60	1,9	3,52
16.11.2006	126	2,10	1,76	1,80	1,6	5,30
17.11.2006	130	2,10	1,86	1,70	1,6	5,49
18.11.2006	110	2,20	1,98	1,60	1,6	7,10
19.11.2006	50	2,10	1,76	1,60	1,5	6,55
20.11.2006	49	2,60	1,63	1,80	1,7	3,85
21.11.2006	63	0,62	1,75	1,70	0,9	4,32
22.11.2006	66	2,10	1,78	1,70	1,6	4,63
23.11.2006	57	2,30	1,93	1,76	1,6	5,13
24.11.2006	44	2,20	1,96	1,85	1,6	4,18
25.11.2006	69	2,10	1,68	1,85	1,5	4,22
26.11.2006	70	2,40	1,98	1,76	1,7	5,26
27.11.2006	46	2,34	1,69	1,62	1,6	5,60
28.11.2006	59	2,20	2,60	1,84	1,8	5,89
29.11.2006	52	2,10	2,10	1,96	1,6	5,10
30.11.2006	44	2,40	2,12	1,86	1,8	4,18
01.12.2006	31	2,20	2,48	1,88	1,8	4,18
02.12.2006	33	2,20	2,56	2,10	1,8	4,10
03.12.2006	60	2,10	2,53	2,30	1,8	4,80
04.12.2006	43	2,10	2,41	1,76	1,7	4,46
05.12.2006	70	2,07	2,21	1,84	1,7	4,76
06.12.2006	35	2,11	1,99	1,92	1,6	3,82
07.12.2006	62	1,57	2,12	2,20	1,5	4,11
08.12.2006	109	1,62	2,13	1,76	1,4	4,51
09.12.2006	70	2,10	2,56	1,92	1,7	4,44
10.12.2006	45	2,40	2,48	1,92	1,8	5,21
11.12.2006	110	2,46	1,78	1,86	1,7	5,16
12.12.2006	130	2,65	1,98	1,85	1,9	5,88
13.12.2006	102	2,75	1,48	1,56	1,8	3,21
14.12.2006	82	2,24	1,76	1,72	1,7	4,29
15.12.2006	98	2,18	1,96	2,10	1,7	2,37

Table B.21. Line III effluent characterization results conducted between 01.11 – 15.12 / 2006

Date	COD_{eff-I}	NO_3-$N_{eff-III}$	$DO_{inf-III}$	$DO_{recyle-III}$	Total $EA_{inf-III}$	PO_4-$P_{eff-III}$
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
01.11.2006	110	2,89	1,98	1,89	2,0	1,76
02.11.2006	95	1,22	2,16	2,22	1,2	1,85
03.11.2006	120	0,72	2,05	2,44	1,0	1,60
04.11.2006	83	0,23	2,26	2,58	0,9	1,32
05.11.2006	75	0,56	2,45	2,58	1,1	2,10
06.11.2006	71	0,20	2,38	2,15	0,8	2,30
07.11.2006	37	0,20	1,88	2,29	0,8	1,86
08.11.2006	32	0,52	1,94	1,92	0,9	1,01
09.11.2006	33	0,72	2,16	2,28	1,0	1,04
10.11.2006	81	2,64	1,96	2,32	1,9	1,96
11.11.2006	53	2,86	2,05	2,36	2,0	1,63
12.11.2006	-	2,10	2,64	2,83	1,8	1,56
13.11.2006	58	0,93	2,48	2,58	1,2	1,44
14.11.2006	62	1,50	2,26	2,71	1,4	1,39
15.11.2006	42	0,58	2,16	2,24	1,0	2,33
16.11.2006	63	1,05	1,94	2,21	1,1	1,56
17.11.2006	59	0,45	2,05	2,09	0,9	2,13
18.11.2006	44	0,30	2,18	1,97	0,8	2,23
19.11.2006	56	0,20	1,94	1,97	0,7	2,13
20.11.2006	65	0,41	1,79	2,21	0,8	2,23
21.11.2006	70	1,38	1,93	2,09	1,3	1,02
22.11.2006	51	0,61	1,96	2,09	0,9	1,39
23.11.2006	54	3,00	2,12	2,16	1,9	2,24
24.11.2006	89	2,98	2,16	2,28	2,0	1,01
25.11.2006	58	3,66	2,12	2,28	2,2	1,88
26.11.2006	55	2,44	2,18	2,16	1,8	1,45
27.11.2006	56	0,80	1,86	1,99	1,0	1,85
28.11.2006	57	0,50	2,86	2,26	1,1	2,47
29.11.2006	65	1,46	2,31	2,41	1,4	2,32
30.11.2006	46	0,64	2,33	2,29	1,1	2,14
01.12.2006	39	0,33	2,73	2,31	1,0	1,75
02.12.2006	45	0,21	2,45	2,26	0,9	2,01
03.12.2006	48	0,23	2,78	2,24	0,9	2,04
04.12.2006	54	0,32	2,25	2,16	0,9	2,32
05.12.2006	92	0,23	2,43	2,26	0,9	1,37
06.12.2006	82	0,28	2,19	2,36	0,9	1,02
07.12.2006	71	0,35	2,33	2,71	1,0	1,03
08.12.2006	61	0,39	2,34	2,16	0,9	1,52
09.12.2006	56	0,23	2,24	2,36	1,0	1,24
10.12.2006	55	0,16	2,73	2,36	0,9	1,26
11.12.2006	63	0,12	1,96	2,29	0,8	2,25
12.12.2006	32	0,18	2,18	2,28	0,8	1,15
13.12.2006	84	0,54	1,63	1,92	0,8	1,24
14.12.2006	80	0,36	1,94	2,12	0,8	1,39
15.12.2006	53	1,28	2,16	2,46	1,4	0,98

Table B.17. Line I effluent characterization results conducted between 16.12.2006 – 28.01. 2007

Date	COD_{eff-I}	NO_3-N_{eff-I}	DO_{inf-I}	$DO_{recyle-I}$	Total EA_{inf-I}	PO_4-P_{eff-I}
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
16.12.2006	112	1,93	1,98	1,86	1,5	3,34
17.12.2006	100	1,85	1,78	1,90	1,5	3,56
18.12.2006	115	1,16	2,21	1,60	1,2	5,12
19.12.2006	101	1,26	2,21	1,90	1,3	6,28
20.12.2006	104	1,36	2,48	1,65	1,3	5,00
21.12.2006	100	1,40	2,26	1,76	1,3	4,21
22.12.2006	130	1,14	2,34	1,58	1,2	4,10
23.12.2006	134	1,41	2,46	1,56	1,4	4,00
24.12.2006	110	1,25	1,85	1,86	1,2	4,56
25.12.2006	102	0,85	1,96	1,96	1,1	6,28
26.12.2006	110	0,76	2,25	1,54	1,0	4,19
27.12.2006	109	0,90	2,24	1,62	1,1	3,44
28.12.2006	100	0,88	2,56	1,94	1,2	4,76
29.12.2006	120	1,65	2,14	1,85	1,4	3,18
30.12.2006	110	1,25	2,23	1,84	1,3	3,73
31.12.2006	100	1,52	1,89	1,86	1,3	3,84
01.01.2007	80	1,48	1,76	1,88	1,3	3,74
02.01.2007	65	1,24	1,85	1,76	1,2	4,58
03.01.2007	72	1,14	1,98	1,74	1,2	5,48
04.01.2007	100	1,59	2,15	1,78	1,4	6,88
05.01.2007	120	1,24	2,18	1,81	1,3	6,23
06.01.2007	120	1,26	2,32	1,84	1,3	7,11
07.01.2007	110	0,96	1,96	1,75	1,1	4,23
08.01.2007	82	0,71	1,76	1,76	0,9	3,86
09.01.2007	84	1,42	1,94	1,68	1,2	3,20
10.01.2007	74	1,36	1,78	1,69	1,2	1,12
11.01.2007	130	1,89	1,98	1,71	1,5	1,17
12.01.2007	128	1,96	1,66	1,74	1,5	1,86
13.01.2007	56	1,20	1,78	1,86	1,2	1,05
14.01.2007	80	1,60	1,96	1,68	1,4	2,10
15.01.2007	85	0,46	1,87	1,76	0,8	2,70
16.01.2007	123	0,51	1,98	1,89	0,9	2,74
17.01.2007	105	1,76	2,23	1,85	1,6	1,01
18.01.2007	101	1,66	2,15	1,76	1,5	2,47
19.01.2007	114	1,42	2,25	1,88	1,4	1,09
20.01.2007	98	1,24	2,23	1,92	1,3	1,69
21.01.2007	85	1,23	2,41	1,66	1,3	2,65
22.01.2007	117	1,12	2,31	1,96	1,3	3,24
23.01.2007	111	1,82	2,25	1,78	1,6	3,04
24.01.2007	100	0,36	2,21	1,77	0,9	1,04
25.01.2007	121	0,94	1,96	1,89	1,1	1,16
26.01.2007	81	0,88	1,98	1,98	1,1	1,23
27.01.2007	120	1,19	1,75	1,65	1,2	1,56
28.01.2007	98	1,02	1,88	1,76	1,1	1,84

Table B.18. Line I effluent characterization results conducted between 29.01 – 25.02 / 2007

Date	COD_{eff-I}	NO_3-N_{eff-I}	DO_{inf-I}	$DO_{recyle-I}$	Total EA_{inf-I}	PO_4-P_{eff-I}
	mg/L	mg/L	mg/L	mg/L	$mgNO_3$-N/L	mg/L
29.01.2007	74	1,16	1,96	1,89	1,2	2,20
30.01.2007	106	1,06	1,76	1,83	1,1	4,11
31.01.2007	92	1,24	1,95	1,81	1,2	5,04
01.02.2007	117	1,45	1,96	1,86	1,3	4,89
02.02.2007	96	2,19	1,87	1,86	1,6	2,64
03.02.2007	81	2,34	1,88	1,85	1,7	2,56
04.02.2007	95	2,10	1,78	1,75	1,6	2,45
05.02.2007	119	2,44	1,76	1,78	1,8	1,95
06.02.2007	100	1,73	1,74	1,89	1,4	3,14
07.02.2007	112	1,21	1,89	1,76	1,2	4,40
08.02.2007	121	1,01	1,88	1,80	1,1	3,70
09.02.2007	137	1,84	1,96	1,85	1,5	1,12
10.02.2007	141	1,46	1,98	1,86	1,3	2,74
11.02.2007	123	1,80	1,56	1,75	1,4	3,21
12.02.2007	113	0,34	1,65	1,76	0,7	3,03
13.02.2007	116	1,41	1,68	1,80	1,3	2,32
14.02.2007	143	0,81	1,76	1,66	0,9	2,13
15.02.2007	70	0,70	1,78	1,90	0,9	1,87
16.02.2007	126	1,03	1,87	1,60	1,1	4,48
17.02.2007	111	1,09	1,98	1,72	1,1	4,12
18.02.2007	115	1,32	1,89	1,76	1,2	3,85
19.02.2007	133	1,70	1,78	1,75	1,4	3,27
20.02.2007	91	0,63	1,76	1,86	0,9	2,17
21.02.2007	98	1,07	1,85	1,76	1,1	1,82
22.02.2007	117	1,10	2,12	1,74	1,2	2,24
23.02.2007	118	1,26	2,21	1,75	1,3	2,40
24.02.2007	121	1,56	2,14	1,76	1,4	2,74
25.02.2007	120	1,80	2,12	1,76	1,5	2,68

Table B.19. Line I effluent characterization results conducted between 26.02 – 31.03 / 2007

Date	COD_{eff-I}	NO_3-N_{eff-I}	DO_{inf-I}	$DO_{recyle-I}$	Total EA_{inf-I}	PO_4-P_{eff-I}
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
26.02.2007	104	0,46	1,89	1,76	0,9	2,74
27.02.2007	107	0,87	1,76	1,85	1,0	2,73
28.02.2007	143	1,10	1,89	1,74	1,1	2,13
01.03.2007	108	1,23	2,14	1,78	1,3	2,39
02.03.2007	81	1,34	2,16	1,76	1,3	1,89
03.03.2007	97	1,40	2,17	1,85	1,4	1,78
04.03.2007	110	1,20	2,17	1,86	1,3	3,85
05.03.2007	116	0,47	2,16	1,68	0,9	5,30
06.03.2007	166	0,53	1,89	1,76	0,9	5,43
07.03.2007	110	1,21	1,76	1,79	1,2	2,56
08.03.2007	130	1,76	1,75	1,92	1,5	1,44
09.03.2007	132	1,22	1,87	1,96	1,2	2,71
10.03.2007	130	1,34	1,76	1,86	1,2	2,33
11.03.2007	110	1,20	1,89	1,88	1,2	2,46
12.03.2007	128	1,10	1,98	1,84	1,2	5,46
13.03.2007	112	1,29	1,75	1,78	1,2	3,85
14.03.2007	110	0,20	1,76	1,87	0,7	3,68
15.03.2007	98	1,30	1,78	1,65	1,2	3,58
16.03.2007	102	1,28	2,12	1,87	1,3	2,65
17.03.2007	120	1,72	2,16	1,77	1,5	3,78
18.03.2007	105	1,28	1,84	1,78	1,2	4,01
19.03.2007	98	0,60	1,98	1,86	1,0	4,48
20.03.2007	106	0,85	1,78	1,81	1,0	2,93
21.03.2007	121	1,01	1,76	1,85	1,1	5,73
22.03.2007	104	1,14	1,85	1,87	1,2	6,38
23.03.2007	110	1,60	1,84	1,86	1,3	5,12
24.03.2007	100	1,55	1,96	1,87	1,3	4,86
25.03.2007	112	1,25	1,94	1,96	1,2	4,98
26.03.2007	120	1,19	1,91	1,77	1,2	5,74
27.03.2007	110	1,14	1,78	1,68	1,1	5,71
28.03.2007	120	1,08	1,79	1,76	1,1	7,23
29.03.2007	110	1,41	1,84	1,95	1,3	7,76
30.03.2007	110	1,32	1,75	1,80	1,2	7,83
31.03.2007	118	1,26	1,96	1,73	1,2	7,88

Table B.20. Line I effluent characterization results conducted between 26.02 – 31.03 / 2007

Date	COD_{eff-I}	NO_3-N_{eff-I}	DO_{inf-I}	$DO_{recyle-I}$	Total EA_{inf-I}	PO_4-P_{eff-I}
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
26.02.2007	104	0,46	1,89	1,76	0,9	2,74
27.02.2007	107	0,87	1,76	1,85	1,0	2,73
28.02.2007	143	1,10	1,89	1,74	1,1	2,13
01.03.2007	108	1,23	2,14	1,78	1,3	2,39
02.03.2007	81	1,34	2,16	1,76	1,3	1,89
03.03.2007	97	1,40	2,17	1,85	1,4	1,78
04.03.2007	110	1,20	2,17	1,86	1,3	3,85
05.03.2007	116	0,47	2,16	1,68	0,9	5,30
06.03.2007	166	0,53	1,89	1,76	0,9	5,43
07.03.2007	110	1,21	1,76	1,79	1,2	2,56
08.03.2007	130	1,76	1,75	1,92	1,5	1,44
09.03.2007	132	1,22	1,87	1,96	1,2	2,71
10.03.2007	130	1,34	1,76	1,86	1,2	2,33
11.03.2007	110	1,20	1,89	1,88	1,2	2,46
12.03.2007	128	1,10	1,98	1,84	1,2	5,46
13.03.2007	112	1,29	1,75	1,78	1,2	3,85
14.03.2007	110	0,20	1,76	1,87	0,7	3,68
15.03.2007	98	1,30	1,78	1,65	1,2	3,58
16.03.2007	102	1,28	2,12	1,87	1,3	2,65
17.03.2007	120	1,72	2,16	1,77	1,5	3,78
18.03.2007	105	1,28	1,84	1,78	1,2	4,01
19.03.2007	98	0,60	1,98	1,86	1,0	4,48
20.03.2007	106	0,85	1,78	1,81	1,0	2,93
21.03.2007	121	1,01	1,76	1,85	1,1	5,73
22.03.2007	104	1,14	1,85	1,87	1,2	6,38
23.03.2007	110	1,60	1,84	1,86	1,3	5,12
24.03.2007	100	1,55	1,96	1,87	1,3	4,86
25.03.2007	112	1,25	1,94	1,96	1,2	4,98
26.03.2007	120	1,19	1,91	1,77	1,2	5,74
27.03.2007	110	1,14	1,78	1,68	1,1	5,71
28.03.2007	120	1,08	1,79	1,76	1,1	7,23
29.03.2007	110	1,41	1,84	1,95	1,3	7,76
30.03.2007	110	1,32	1,75	1,80	1,2	7,83
31.03.2007	118	1,26	1,96	1,73	1,2	7,88

Table B.21. Line III effluent characterization results conducted between 01.11 – 15.12 / 2006

Date	COD_{eff-I}	NO_3-$N_{eff-III}$	$DO_{inf-III}$	$DO_{recyle-III}$	Total $EA_{inf-III}$	PO_4-$P_{eff-III}$
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
01.11.2006	110	2,89	1,98	1,89	2,0	1,76
02.11.2006	95	1,22	2,16	2,22	1,2	1,85
03.11.2006	120	0,72	2,05	2,44	1,0	1,60
04.11.2006	83	0,23	2,26	2,58	0,9	1,32
05.11.2006	75	0,56	2,45	2,58	1,1	2,10
06.11.2006	71	0,20	2,38	2,15	0,8	2,30
07.11.2006	37	0,20	1,88	2,29	0,8	1,86
08.11.2006	32	0,52	1,94	1,92	0,9	1,01
09.11.2006	33	0,72	2,16	2,28	1,0	1,04
10.11.2006	81	2,64	1,96	2,32	1,9	1,96
11.11.2006	53	2,86	2,05	2,36	2,0	1,63
12.11.2006	-	2,10	2,64	2,83	1,8	1,56
13.11.2006	58	0,93	2,48	2,58	1,2	1,44
14.11.2006	62	1,50	2,26	2,71	1,4	1,39
15.11.2006	42	0,58	2,16	2,24	1,0	2,33
16.11.2006	63	1,05	1,94	2,21	1,1	1,56
17.11.2006	59	0,45	2,05	2,09	0,9	2,13
18.11.2006	44	0,30	2,18	1,97	0,8	2,23
19.11.2006	56	0,20	1,94	1,97	0,7	2,13
20.11.2006	65	0,41	1,79	2,21	0,8	2,23
21.11.2006	70	1,38	1,93	2,09	1,3	1,02
22.11.2006	51	0,61	1,96	2,09	0,9	1,39
23.11.2006	54	3,00	2,12	2,16	1,9	2,24
24.11.2006	89	2,98	2,16	2,28	2,0	1,01
25.11.2006	58	3,66	2,12	2,28	2,2	1,88
26.11.2006	55	2,44	2,18	2,16	1,8	1,45
27.11.2006	56	0,80	1,86	1,99	1,0	1,85
28.11.2006	57	0,50	2,86	2,26	1,1	2,47
29.11.2006	65	1,46	2,31	2,41	1,4	2,32
30.11.2006	46	0,64	2,33	2,29	1,1	2,14
01.12.2006	39	0,33	2,73	2,31	1,0	1,75
02.12.2006	45	0,21	2,45	2,26	0,9	2,01
03.12.2006	48	0,23	2,78	2,24	0,9	2,04
04.12.2006	54	0,32	2,25	2,16	0,9	2,32
05.12.2006	92	0,23	2,43	2,26	0,9	1,37
06.12.2006	82	0,28	2,19	2,36	0,9	1,02
07.12.2006	71	0,35	2,33	2,71	1,0	1,03
08.12.2006	61	0,39	2,34	2,16	0,9	1,52
09.12.2006	56	0,23	2,24	2,36	1,0	1,24
10.12.2006	55	0,16	2,73	2,36	0,9	1,26
11.12.2006	63	0,12	1,96	2,29	0,8	2,25
12.12.2006	32	0,18	2,18	2,28	0,8	1,15
13.12.2006	84	0,54	1,63	1,92	0,8	1,24
14.12.2006	80	0,36	1,94	2,12	0,8	1,39
15.12.2006	53	1,28	2,16	2,46	1,4	0,98

Table B.22. Line I effluent characterization results conducted between 16.12.2006-28.01.2007

Date	COD_{eff-I}	NO_3-$N_{eff-III}$	$DO_{inf-III}$	$DO_{recyle-III}$	Total $EA_{inf-III}$	PO_4-$P_{eff-III}$
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
16.12.2006	34	1,73	2,18	2,29	1,5	1,11
17.12.2006	46	1,44	1,96	2,34	1,4	1,16
18.12.2006	41	4,13	2,43	1,97	2,6	0,98
19.12.2006	110	0,47	2,43	2,34	1,0	0,86
20.12.2006	58	3,10	2,73	2,23	2,2	1,88
21.12.2006	85	3,36	2,49	2,16	2,3	2,16
22.12.2006	46	3,10	2,57	1,94	2,2	1,78
23.12.2006	30	3,12	2,71	1,92	2,2	1,84
24.12.2006	44	3,21	2,04	2,29	2,2	1,86
25.12.2006	45	0,44	2,16	2,41	1,0	1,87
26.12.2006	75	0,27	2,48	1,89	0,9	1,78
27.12.2006	54	0,44	2,46	1,99	0,9	1,13
28.12.2006	79	0,41	2,82	2,39	1,1	1,84
29.12.2006	36	0,53	2,35	2,28	1,0	2,04
30.12.2006	46	0,46	2,45	2,26	1,0	2,36
31.12.2006	44	0,56	2,08	2,29	1,0	1,06
01.01.2007	55	1,12	1,94	2,31	1,2	1,05
02.01.2007	56	1,24	2,04	2,16	1,2	0,86
03.01.2007	72	1,06	2,18	2,14	1,2	1,12
04.01.2007	125	0,27	2,37	2,19	0,9	1,13
05.01.2007	99	0,42	2,40	2,23	1,0	1,16
06.01.2007	120	0,89	2,55	2,26	1,2	2,35
07.01.2007	110	0,42	2,16	2,15	0,9	2,13
08.01.2007	78	0,46	1,94	2,16	0,9	2,13
09.01.2007	122	0,99	2,13	2,07	1,1	2,21
10.01.2007	102	0,98	1,96	2,08	1,1	2,36
11.01.2007	120	0,21	2,18	2,10	0,8	2,04
12.01.2007	114	0,52	1,83	2,14	0,9	2,21
13.01.2007	97	0,42	1,96	2,29	0,9	2,11
14.01.2007	98	0,44	2,16	2,07	0,9	2,13
15.01.2007	96	0,43	2,06	2,16	0,9	2,01
16.01.2007	109	0,44	2,18	2,32	1,0	2,03
17.01.2007	126	0,79	2,28	1,76	1,1	2,44
18.01.2007	137	0,78	2,37	2,16	1,1	2,56
19.01.2007	110	1,18	2,48	2,31	1,3	1,92
20.01.2007	120	1,20	2,45	2,36	1,3	1,70
21.01.2007	110	1,13	2,46	1,86	1,3	1,60
22.01.2007	113	1,26	2,54	2,41	1,4	1,85
23.01.2007	131	0,34	2,48	2,19	0,9	1,87
24.01.2007	126	1,08	2,43	2,18	1,3	1,23
25.01.2007	110	0,12	2,16	2,32	0,8	1,18
26.01.2007	92	0,16	2,18	2,11	0,8	0,74
27.01.2007	110	0,38	1,93	2,03	0,8	1,65
28.01.2007	105	0,44	2,07	2,16	0,9	1,55

Table B.23. Line III effluent characterization results conducted between 29.01 – 25.02 / 2007

Date	COD_{eff-I}	NO_3-$N_{eff-III}$	$DO_{inf-III}$	$DO_{recyle-III}$	Total $EA_{inf-III}$	PO_4-$P_{eff-III}$
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
29.01.2007	122	0,16	2,16	2,32	0,8	1,89
30.01.2007	110	1,07	1,94	2,25	1,2	1,33
31.01.2007	95	0,19	2,15	2,23	0,8	1,02
01.02.2007	182	0,35	2,16	2,29	0,9	2,09
02.02.2007	115	0,19	2,06	2,29	0,8	2,45
03.02.2007	112	0,21	2,07	2,28	0,8	2,88
04.02.2007	110	0,11	1,96	2,15	0,7	2,21
05.02.2007	130	0,27	1,94	2,16	0,8	2,06
06.02.2007	120	0,15	1,91	2,32	0,8	1,74
07.02.2007	113	0,18	2,12	2,16	0,8	1,68
08.02.2007	169	0,15	2,07	2,21	0,8	1,50
09.02.2007	118	0,27	2,16	2,28	0,9	1,76
10.02.2007	107	0,41	2,18	2,29	0,9	2,23
11.02.2007	104	0,06	1,72	2,15	0,7	2,30
12.02.2007	128	0,19	1,82	2,16	0,7	2,21
13.02.2007	123	0,13	1,85	2,21	0,7	2,36
14.02.2007	137	0,23	1,94	1,96	0,7	2,11
15.02.2007	121	0,21	2,23	2,34	0,8	2,11
16.02.2007	98	0,15	2,06	1,97	0,7	2,23
17.02.2007	116	0,26	2,18	2,12	0,8	1,86
18.02.2007	120	0,33	2,08	2,16	0,9	2,21
19.02.2007	102	0,40	1,96	2,15	0,9	2,62
20.02.2007	93	0,15	1,94	2,29	0,8	0,97
21.02.2007	95	0,19	2,26	2,16	0,8	1,08
22.02.2007	108	0,23	2,33	2,14	0,9	1,88
23.02.2007	110	0,22	2,43	2,15	0,9	2,16
24.02.2007	43	1,26	2,35	2,16	1,3	2,04
25.02.2007	48	0,86	2,33	2,16	1,1	1,86

Table B.24. Line III effluent characterization results conducted between 26.02 – 31.03 / 2007

Date	COD_{eff-I}	NO_3-$N_{eff-III}$	$DO_{inf-III}$	$DO_{recyle-III}$	Total $EA_{inf-III}$	PO_4-$P_{eff-III}$
	mg/L	mg/L	mg/L	mg/L	mgNO_3-N/L	mg/L
26.02.2007	56	0,75	2,08	2,16	1,1	1,92
27.02.2007	56	0,65	1,94	2,30	1,0	1,97
28.02.2007	89	0,23	2,12	2,14	0,8	1,01
01.03.2007	84	0,44	2,35	2,19	1,0	1,95
02.03.2007	92	0,39	2,38	2,16	0,9	0,81
03.03.2007	88	0,42	2,39	2,28	1,0	0,92
04.03.2007	98	0,34	2,39	2,11	0,9	0,56
05.03.2007	88	0,28	2,38	2,07	0,9	1,04
06.03.2007	146	0,30	2,08	2,16	0,8	1,18
07.03.2007	138	0,43	1,94	2,20	0,9	1,71
08.03.2007	81	0,61	1,93	2,36	1,0	0,88
09.03.2007	100	0,22	2,06	2,41	0,8	1,50
10.03.2007	110	0,26	1,94	2,11	0,8	1,22
11.03.2007	63	0,32	2,08	2,31	0,9	1,34
12.03.2007	113	0,35	2,18	2,26	0,9	1,26
13.03.2007	90	0,34	2,23	2,22	0,9	1,15
14.03.2007	112	0,50	2,26	2,30	0,9	1,16
15.03.2007	93	0,57	1,96	2,03	0,9	1,02
16.03.2007	106	0,60	2,33	2,30	1,1	0,82
17.03.2007	65	0,40	2,38	2,18	0,9	1,80
18.03.2007	72	0,26	2,02	2,19	0,8	1,65
19.03.2007	113	0,85	2,18	2,22	1,1	1,86
20.03.2007	110	1,16	1,96	2,23	1,3	1,84
21.03.2007	86	1,59	1,94	2,28	1,4	1,82
22.03.2007	86	1,07	2,04	2,30	1,2	2,11
23.03.2007	95	1,82	2,02	2,29	1,5	2,23
24.03.2007	97	1,55	2,16	2,16	1,3	2,41
25.03.2007	102	1,23	2,13	2,41	1,3	2,44
26.03.2007	89	1,42	2,10	2,18	1,4	2,23
27.03.2007	75	1,34	1,96	2,23	1,3	2,33
28.03.2007	100	1,27	1,97	2,16	1,3	1,21
29.03.2007	120	1,26	2,24	2,40	1,3	1,16
30.03.2007	110	1,28	1,87	2,12	1,3	1,14
31.03.2007	98	1,30	2,18	1,89	1,3	1,33

Table B.25. Anaerobic P release bath-scale experimental results conducted in 07.01.2007

Date:	07.01.2007			
Line:	I			
MLSS:	mg/L	2940	2950	2960
LOI:	%	76,1	76,2	75,6
MLVSS:	g/l	2,2	2,2	2,2

Date:	07.01.2007			
Line:	IIII			
MLSS:	mg/L	3780	3785	3770
LOI:	%	77,1	77,2	77,1
MLVSS:	g/l	2,9	2,9	2,9

Time Minutes	**NO_3-N-I**	**PO_4-P-I**
	mg/L	**mg/L**
0	0,1	1,2
1	18,2	32,1
30	17,3	19,8
60	15,6	14,8
90	13,2	12,8
120	9,8	8,6
180	7,2	7,5
210	6,4	4,4

Time Minutes	**NO_3-N-III**	**PO_4-P-III**
	mg/L	**mg/L**
0	0,1	1,1
1	24,1	62,4
30	18,4	42,4
60	15,1	24,6
90	12,3	9,1
120	8,4	4,2
180	4,6	2,3
210	0,9	1,2

Table B.26. Anoxic P uptake – denitrification bath-scale experimental results conducted in 07.01.2007

Date:	07.01.2007			
Line:	I			
MLSS:	mg/L	2940	2950	2960
LOI:	%	76,1	76,2	75,6
MLVSS:	g/l	2,2	2,2	2,2

Date:	07.01.2007			
Line:	IIII			
MLSS:	mg/L	3780	3785	3770
LOI:	%	77,1	77,2	77,1
MLVSS:	g/l	2,9	2,9	2,9

Time Minutes	**NO_3-N-I**	**PO_4-P-I**
	mg/L	**mg/L**
0	0,1	1,2
1	18,2	32,1
30	17,3	19,8
60	15,6	14,8
90	13,2	12,8
120	9,8	8,6
180	7,2	7,5
210	6,4	4,4

Time Minutes	**NO_3-N-III**	**PO_4-P-III**
	mg/L	**mg/L**
0	0,1	1,1
1	24,1	62,4
30	18,4	42,4
60	15,1	24,6
90	12,3	9,1
120	8,4	4,2
180	4,6	2,3
210	0,9	1,2

Table B.27. Aerobic P uptake bath-scale experimental results conducted in 07.01.2007

Date:	07.01.2007			
Line:	I			
MLSS:	mg/L	2940	2950	2960
LOI:	%	76,1	76,2	75,6
MLVSS:	g/l	2,2	2,2	2,2

Time Minutes	**PO_4-P-I**
	mg/L
1	32,10
30	23,40
60	14,40
90	8,50
120	7,30
180	4,10
210	3,20

Date:	07.01.2007			
Line:	III			
MLSS:	mg/L	3780	3785	3770
LOI:	%	77,1	77,2	77,1
MLVSS:	g/l	2,9	2,9	2,9

Time Minutes	**PO_4-P-III**
	mg/L
1	65,00
30	44,60
60	20,60
90	8,40
120	3,80
180	1,80
210	0,75

Table C.28. Anaerobic P release bath-scale experimental results conducted in 22.02.2007

Date:	22.02.2007			
Line:	I			
MLSS:	mg/L	2360	2380	2330
LOI:	%	74,75	74,56	75,48
MLVSS:	g/l	1,8	1,8	1,8

Date:	22.02.2007			
Line:	IIII			
MLSS:	mg/L	2460	2450	2480
LOI:	%	74,75	74,56	74,48
MLVSS:	g/l	1,8	1,8	1,8

Time minutes	**PO_4-P-I**	**sCOD-I**
	mg/L	**mg/L**
0	0,16	58,6
1	4,5	658,4
30	13,82	616,1
60	26,8	576,2
90	36,12	534
120	36,18	534,6
180	35,8	532,5
210	36,21	535,1

Time minutes	**PO_4-P-III**	**sCOD-III**
	mg/L	**mg/L**
0	0,22	58,6
1	1,2	664
30	18,16	604
60	36,52	542
90	46,12	481
120	46,17	486
180	46,15	476
210	46,08	480

Table B.29. Anoxic P uptake – denitrification bath-scale experimental results conducted in 22.02.2007

Date:	22.02.2007			
Line:	I			
MLSS:	mg/L	2120	2130	2105
LOI:	%	74,75	74,56	74,49
MLVSS:	g/m^3	1,6	1,6	1,6

Date:	22.02.2007			
Line:	III			
MLSS:	mg/L	2160	2150	2168
LOI:	%	74,48	74,36	74,46
MLVSS:	g/m^3	1,6	1,6	1,6

Time Minutes	**NO_3-N-I**	**PO_4-P-I**
	mg/L	**mg/L**
0	0,12	1,20
1	18,24	25,60
30	17,10	16,20
60	15,50	14,20
90	13,50	11,40
120	10,20	9,14
180	8,14	7,40
210	6,74	4,70

Time Minutes	**NO_3-N-III**	**PO_4-P-III**
	mg/L	**mg/L**
0	0,13	1,20
1	18,40	36,42
30	16,50	25,42
60	13,40	15,30
90	11,30	5,21
120	9,20	3,82
180	6,85	2,28
210	4,40	1,26

Table B.30. Aerobic P uptake bath-scale experimental results conducted in 22.02.2007

Date:	22.02.2007			
Line:	I			
MLSS:	mg/L	2120	2115	2120
LOI:	%	74,75	74,85	74,36
MLVSS:	g/m^3	1,6	1,6	1,6

Time Minutes	**PO_4-P-I**
	mg/L
1	26,11
30	21,18
60	13,64
90	7,90
120	6,26
180	3,58
210	2,10

Date:	22.02.2007			
Line:	III			
MLSS:	mg/L	2140	2130	2160
LOI:	%	74,51	74,83	74,18
MLVSS:	g/m^3	1,6	1,6	1,6

Time Minutes	**PO_4-P-III**
	mg/L
1	35,60
30	28,64
60	16,48
90	5,82
120	2,86
180	1,24
210	0,85

Table B.31. Anaerobic P release bath-scale experimental results conducted in 07.03.2007

Date:	07.03.2007			
Line:	I			
MLSS:	mg/L	3650	3640	3660
LOI:	%	76,1	76,2	76,3
MLVSS:	g/L	2,8	2,8	2,8

Time minutes	**PO4-P-I**	**sCOD-I**
	mg/L	**mg/L**
0	0,4	70,2
1	2,6	656,1
30	14,4	606,1
60	26,1	554,1
90	37,5	501,1
120	37,6	500,0
180	37,5	503,3
210	37,4	510,6

Date:	07.03.2007			
Line:	IIII			
MLSS:	mg/L	3750	3760	3740
LOI:	%	74,0	74,1	74,0
MLVSS:	g/L	2,8	2,8	2,8

Time minutes	**PO4-P-III**	**sCOD-III**
	mg/L	**mg/L**
0	0,5	85,1
1	1,8	658,6
30	28,6	558,6
60	56,2	460,6
90	68,0	376
120	67,6	366
180	67,5	368
210	68,1	372

Table B.32. Anoxic P uptake – denitrification bath-scale experimental results conducted in 07.03.2007

Date:	07.03.2007			
Line:	I			
MLSS:	mg/L	2940	2950	2960
LOI:	%	76,1	76,2	75,6
MLVSS:	g/m^3	2,2	2,2	2,2

Time Minutes	**NO_3-N-I**	**PO_4-P-I**
	mg/L	**mg/L**
0	1,1	1,2
1	19,6	27,6
30	18,4	18,1
60	16,3	14,1
90	13,6	11,6
120	10,8	7,5
180	8,8	5,8
210	7,6	3,6

Date:	07.03.2007			
Line:	IIII			
MLSS:	mg/L	3780	3785	3770
LOI:	%	77,1	77,2	77,1
MLVSS:	g/m^3	2,9	2,9	2,9

Time Minutes	**NO_3-N-III**	**PO_4-P-III**
	mg/L	**mg/L**
0	0,1	1,1
1	23,4	60,3
30	18,4	40,1
60	15,1	21,8
90	12,3	7,6
120	8,4	2,2
180	4,6	1,1
210	1,9	0,4

Table C.31. Aerobic P uptake bath-scale experimental results conducted in 07.03.2007

Date:	07.03.2007			
Line:	I			
MLSS:	mg/L	2940	2950	2960
LOI:	%	76,1	76,2	75,6
MLVSS:	g/m^3	2,2	2,2	2,2

Time Minutes	**PO_4-P-I**
	mg/L
1	27,80
30	22,10
60	13,30
90	8,40
120	7,10
180	3,80
210	3,30

Date:	07.03.2007			
Line:	IIII			
MLSS:	mg/L	3780	3785	3770
LOI:	%	77,1	77,2	77,1
MLVSS:	g/m^3	2,9	2,9	2,9

Time Minutes	**PO_4-P-III**
	mg/L
1	60,10
30	42,20
60	21,60
90	9,10
120	3,70
180	2,20
210	0,80