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Simultaneous Organic and Nutrients Removal in a Mineral Filled Pilot scale Trickling filter treating Brewery wastewater

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Table of Contents

Preface

This thesis represents a culmination of work and learning that has taken place over a period of four years (2010 - 2014). It is produced as cumulative dissertation which consists of an overview and individual publications which are recently published in peer reviewed international scientific journals, manuscripts, and supplementary chapters incorporated in the bound copy. This research was funded by Landesgraduiertenförderung M-V. Certainly, I would have never reached the point of finishing my dissertation without the help and support of others.

The issues that are discussed in the introduction chapter of the present desertation includes: the demand for production of beer and internal and external drives of the brewing sectors for environmental protection measures.Wastewater generation and environmental impacts of untreated brewery wastewater is also dicussed breifly. This is followed by the discussion of the advantages of trickling filter process over the other wastewater treatment technologies for the treatment of brewery wastewater that is proposed and investigated in this PhD research. Basis of trickling filter process and approaches of trickling filter modeling is the next chapter following the first introduction chapter. The subsequent chapters in this cumulative dissertation were separately published (chapters 3, 5 & 7) and contain supplementary yet unpublished work (chapters 4 & 6) that are coupled each other by discussing the performance of the bio reactor for the removal of organics and nutrients during phase one and phase two operation of the trickling filter using synthetic and industrial brewery wastewater respectively. Analysis of the kinetic behavior of the trickling filter using existing steady state trickling filter models is also included in these chapters.Chapter 8 discusses the cost savings by the trickling fiter when compared to other conventional wastewater treatment system. The summary of the major findings and general conclusions of the present study is discussed in chapter 9. Beside this in this chapter area for future research that further improves the efficiency of the trickling filter with respect to nutrient removal is recommended.

List of tables

List of figures

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I am deeply grateful to my parents: mom, brothers and sisters that were encouraged me to start my PhD study. Without my parents: Mom tiruwork , sisters: Rosa, Tsege, Tizita, Hiwot, Jerusalem, and brothers, Melaku, Tekalegn and Endalkachew, endless love, support and encouragement, I could not have completed my Ph.D study. Last but foremost I want to thank my creator God and his mother. I can not count the blessings from him. Thinking of his presence was the first and foremost drive to move in life.

Publications

To a large extent this thesis is based on the following publications

Habte Lemji* and Hartmut Eckstädt. A pilot scale trickling filter with pebble gravel as media and its Performance to remove chemical oxygen demand from synthetic brewery waste water. International journal of Biomedicine & Biotechnology; Springer **2013 14 (10):924-933**

Habte Lemji* and Hartmut Eckstädt. Nutrient removal with synthetic brewery wastewater in trickling filter bio film. International journal of microbiology and biotechnology **IJAMBR 2 (2014) 30-42**

Habte Lemji* and Hartmut Eckstädt. Efficiency of a pilot scale trickling filter to treat industrial brewery wastewater: Influence of hydraulic loading. J. Chem. Technol. Biotechnol. **Volume 90, Issue 1, pages 201–207, January 2015**

List of acronyms

AOB	Ammonium oxidizing bacteria
A	Total area of the biofilm [m^2]
Ar	Cross-sectional area (m^2)
Ag	Contact area between gas phase [air] and bulk water [m^2]
$A_gJ_{g,\ i}$	Total flux of a gaseous substance [g/d]
As	Specific surface area of the medium [m^2/m^3]
BNR	Biological nutrient removal
COD	Soluble Chemical Oxygen Demand [mg/l]
C_{in}	Influent concentration [mg/L]
C_{out}	Effluent concentration [mg/L]
CLSM	Confocal laser scanning microscopy
CSBRs	Continuously stirred biofilm reactors
D	Depth of the trickling filter [m]
Di	The diffusion coefficient for substrate i
DEM	Deutsche Mark
DGGE	Denaturating gradient gel electroporosis
D PAO_s	Denitrifying phosphorous accumulating organisms
E	The fraction of BOD removed [%]
EBPR	Enhanced biological phosphorous removal
FISH	Fluorescent in situ hybridization
FFR	French Frank
G	Transfer function
h	The bioreactor length [m]
HLRs	Hydraulic loading rate m^3/m^2
HUF	Hungarian Forint
J_g	The flux from gas to bulk [g/m^2d]
Jg,O_2	Oxygen flux from gas to bulk [g/m^2d]

J_f	Flux from biofilm to bulk [g/m^2d]
K_{20}	Treatability constant
K_b	Bio kinetic parameter [g/m^2 [media area] d]
k_d	Rate constant [m^{-1}]
K_L,O_2	Ogygen mass transfer coeffecient
L	The biofim thickness [m]
L'	Volumetric soluble organic loading [kg/m^3 d]
L"	Surface soluble organic loading [g/m^2 [media area] d]
NOK	Norwegische Krone
m_σ	The mass of the added trace substance
n	Hydraulic flow exponent [0.5]
n_1	The number of substrates considered in the model
NOB	Nitrite oxidizing bacteria
OLRs	Organic loading rate [kg COD/m^3/d]
PCR	Polymerase chain reaction
q	Specific hydraulic loading [m^3/m^2]
Q	Hydraulic loading [m^3/m^2 [c.s. area]d]
R_f	A recycle factor
RNA	Ribonucleic acid
RTDs	Resifence time distribution
So	Influent COD concentration [mg/l]
S'	Effluent COD concentration [mg/l]
S^b	Substrate concentration in the bulk [g/m^3]
$S_{O_2}^b$	Bulk oxygen concentration
S_{in}^b	Influent bulk concentration
S	The concentration in the biofilm the distance (m) from substratum
U_{max}	Bio kinetic parameter [g/m^2 [media area] d]
v	Specific bulk volume [m^3]

V	Bulk water volume [m^3]
V_m	Total volume of filter media [10^3 m^3]
w	BOD loading rate [kg BOD/d]
X	Bacterial concentration
θ	Temperature correction factor [1.035]
λ	Time scaling coefficient
τ	Time constant [non-dimensional or d]
γ	Non-dimensional coefficient for substrate flux into biofilm
G	Transfer function, transfer function matrix
Z	The level in the reactor

1 Introduction

1.1 Background and rationale of the research

Beer is an alcoholic beverage usually made from malted cereal grain (as barley), flavoured with hops, and brewed by slow fermentation (Barth, Roger, 2013). It is produced by the saccharification of starch and fermentation of the resulting sugar. The starch and saccharification enzymes are often derived from malted cereal grains; most commonly malted barley and malted wheat. Beer is the world's most widely consumed alcoholic beverage and is the third-most popular drink overall, after water and tea (Nelson, M., 2005). It is thought by some to be the oldest fermented beverage (Rudgley, R., 1993) (Arnold & John P., 2005) (Joshua, J. Mark, 2011) (World's Best Beers, 2009). Today, the brewing industry is a global business, consisting of several dominant multinational companies and many thousands of smaller producers ranging from brewpubs to regional breweries.

During the production of beer huge wastewater generation from cooling (eg.saccharification cooling, fermentation) and washing units often causes several environmental problems. Substantial improvements has been made in the past, however it has been estimated that approximately 3 to 22 L of waste effluent is generated per liter of beer produced in breweries (Kanagachandran, K., Jayaratne, R.,2006 ; UNEP, ABIWSI fact sheet). Water consumption is divided into 2/3 used in the process and 1/3 in the cleaning operations (Moll, 1991). In the same way, effluent to beer ratio is correlated to beer production. It has been shown that the effluent load is very similar to the water load since none of this water is used to brew beer and most of it ends up as effluent (Perry, 2003).

“Effluent discharge from brewery industry contains high concentration of organic matter and nutrients, therefore it should be treated prior to discharge to surface water bodies so as to decrease the high oxygen demand of the wastewater”. Direct discharge can bring about a rapid deterioration of the physical, chemical, and biological qualities of the receiving water bodies (The Breweries of Europe, 2002; Parawira *et al*., 2005; Al-Rekabil *et al*., 2007).

When industrial effluent high in BOD is discharged to water bodies microorganisms living in the oxygenated water body use dissolved oxygen to oxidatively degrade the organic compounds, releasing energy which is used for growth and reproduction. Populations of these microorganisms tend to increase in proportion to the amount of food available. This microbial metabolism creates an oxygen demand proportional to the amount of organic compounds useful as food. Under some circumstances, microbial metabolism can consume dissolved oxygen faster than atmospheric oxygen can dissolve into the water or the autotrophic community (algae, cyanobacteria and macrophytes) can produce. As a result fish and aquatic insects may die when oxygen is depleted by microbial metabolism (Goldman et al., 1983). Therefore the removal of organic compounds from the wastewater is important to avoid excessive depletion of oxygen in water bodies. Nutrients like nitrogen (N) and phosphorous (P) should also be removed to avoid algal blooms that disturb the ecosystem of the receiving waters (Driessen, W., and Vereijken, T., 2003). Furthermore turbidity and color reduces the penetration of light, which, in turn, affects photosynthesis, there by affecting the primary link in the food chain.

Brewery wastewater contains high ammonia nitrogen which might come from the organically bound nitrogen. The wastewater containing ammonia nitrogen may cause eutrophication (Nixon, 1995) and produce toxic substances if discharged into the aquatic eco-system (Yamamoto, 2003). Ammonia also increases chlorine consumption for water disinfection and industrial circulating water sterilization treatment. It is therefore imperative to remove ammonia nitrogen during the wastewater treatment, which could add to the difficulty and cost of the treatment (Frink, 1967). The removal of ammonia from wastewater has become a worldwide emerging concern because ammonia is toxic to aquatic species and causes eutrophication in natural water environments (Tchobanoglous *et al*., 2003).

The other component of the wastewater is phosphorus. Phosphorus compounds are carried into both ground and surface waters with the wastewater due to their presence in many detergents which are used in the cleaning operation of the brewery. Consequently, they are carried into both ground and surface waters, industrial wastes and storm water (Bartram and Balance, 1996). Orthophosphate or polyphosphate are the primary forms of inorganic phosphate in natural waters (Walker, 2001). Eutrophication of natural water sources is the result of excessive level of phosphorous (DeZuane, 1997). Fish kill will result which is attributed to the severely reduced level of oxygen by the growth of algae. Certain algae, the cyanobacteria, contain hepatotoxins which may act as promoters in hepatocarcinogenesis (Frank, 1996).

An increased investment for environmental protection measures due to a significant increase in environmental awareness by the breweing sectors has taken place during the last years. Important internal drivers for the brewing industry are implementation of environmental management systems (EMS) like ISO 14001 as well as the need for conducting benchmark studies for brewery process optimization. Knowledge about environmental emissions (e.g.effluent quality and quantity) can become management information, which may help to improve the efficiency of in-plant brewery processes (minimize product losses, spill of water and energy). Important external drivers for environmental investments are local legislation and environmental taxation systems (discharge levels). The overall result is a growing interest within the brewing industry in environmental pollution controls systems.

Generally, biological methods are adopted for beer brewery wastewater treatment, reported to perform well in chemical oxygen demand (COD) removal (Ince *et al.,* 2000; Parawira *et al.,* 2005). However, the large amount of energy input in wastewater treatment, especially energy consumed by aeration procedures in aerobic treatment, has been considered as a big problem for many years in wastewater treatment in many countries. For instance in the brewery industry at paonta sahib (H.P), India activated sludge process initially used in 1997 is suffering from high energy requirements for the aeration and inconsistency in achieving the effluent standards (Avinash Kumar Sharda *et al.*, 2013). In addition to this in activated sludge process control of effluent ammonia while still meeting the nutritional requirements of the biomass is an operational challenge.

It has been found that most of the widely used conventional processes are affected by some parameters inside the wastewater such as ammonium nitrogen which also contained in brewery wastewater. Ammonium nitrogen can be removed by raising the pH value and then air-stripping the wastewater. However, this method is sometimes quite ineffective, depending on the operational parameters used and water quality, and the cost is usually prohibitively high (Henry, J.G., Prasad, D., 2000; LI, H.,2000; Zhao, Y., 1999; Zhao, Y., 2001; Zhao, Y. *et al.*, 2000) therefore might not be affordable for low income countries particularly. For instance a number of pollution related studies have confirmed that about 90% of industries including breweries in Ethiopia, Addis Ababa are simply discharging their effluent into near by water bodies, streams and open land without any form of treatment (EEPA, 2006) due to the high cost of pollution control technologies.

On the other hand the feasibility of some of these treatment technologies is limited by sludge handling problem, for example activated sludge in suspended growth process normally exhibits poor settleability, as such, and fixed-film systems that would involve trickling filters, rotating biological contactors, et cetera are recommended (Zurchin, J.P. et al., 1986).

Trickling bio filters and tower filtration technologies are regarded as well-established treatment technologies for municipal wastewater (Metcalf and Eddy, 1991). Unlike the other conventional treatment technologies, trickling filter do not require high investment in mechanical or energy demanding equipment and does not require much human attendance for operation and maintenance of the systems. In trickling filters,which are packed bed reactors the effluent flows downward thus trickling on the surface area of the packed bed particles where as the organic and nutrients are assimilated by the biomass growing on the packed bed media. This is the basic principle underlying the high water treatability of this biological treatment method (Wiki, 1999). It is therefore expected to provide with high-quality effluents using cost-effective technologies (Seaguret et al., 2000). Several factors are important in the function of TFs (trickling filters) such as hydraulic load, filter media characteristics (type, size and surface area) and reactors' dimensions.

When we see the condition in developing countries, most of the countries around the globe are in the time of rapid industrialization. Due to this fact, large numbers of industries are emerging mostly in the urban areas of developing countries. On the other hand almost all of the countries give very little attention to the environmental impact of the wastewater, which is an obvious by product of all industries. The plant designer and supplier normally provide an in-built pollution control system for new industries. The peripheral facilities, like wastewater treatment, suffer due to limited financial resources. In addition, other factors such as lack of experience in operation, management and plant repairs, lack of spare parts, frequent shortage of power/fuel, and lack of end products disposal facilities, social and political reasons contribute to inadequate wastewater management in developing countries. Absence of industrial effluent standards and corresponding legislation for enforcing them are common in developing countries. Environmental impact assessment studies for the industrial growth are commonly not carried out in developing countries. Public awareness against pollution is also at a low profile in developing countries (N.K.Pareek, 1992).

Breweries are a widespread industry in Africa and brewing is intrinsically a water intensive industry. According to the sectoral study and framework analysis conducted in Ethiopia, Ghana, Morocco and Uganda, water consumption and specific use (hl water/hl beer) varies greatly between breweries in the study countries and ranges from 7.2 hl/hl in Uganda to 22 hl/hl in Ethiopia. Therefore, this work includes a detailed study of the performance evaluation of naturally ventilated gravel-filled trickling filters for the treatment of brewery wastewater.

1.2 Objectives of the research

Several researchers have reported on the performance of a trickling filter for the treatment of toxic and volatile organic contaminants, metals and nitrogen removal. However the performance of a trickling filter, for simultaneous organic and nutrient removal from brewery wastewater without a need for special arrangement for nitrogen and phosphorus removal is not investigated so far and therefore this was the aim of this PhD thesis. The objectives of the present thesis entails: (a) The study of the startup behavior of the trickling filter under ambient temperature and pH condition (b) investigation of organics and nutrient removal in the trickling filter system under different operating conditions (c) analysis of bio kinetic behavior of the trickling filter with respect to the removal of organic substances (d) analysis of the cost savings as a result of the employment of trickling filter process instead of other conventional treatment technologies for remediation of brewery wastewater.

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2 Basis of trickling filter and trickling filter modeling

In this chapter of the thesis an overview of the trickling filter's biology, technology and its use in industrial wastewater treatment and fundamental concept on dynamic trickling filter model is discussed. An organic matrix consisting of a complex community of bacteria, algae, fungi and protozoa embedded in organic polymers termed as a biofilm. Biofilter media that, generally impermeable are used as a support for the attachment of the biofilm. The reactants are transported from the bulk liquid to the biofilm by diffusion. The bacteria carry out the desired transformation of the reactants into the biofilm. Trickling filters are one of the typical examples of fixed biofilm reactors.

A trickling filter is a type of water pollution treatment system. It consists of a fixed bed of rocks, lava, coke, gravel, slag, polyurethane foam, sphagnum peat moss, ceramic, or plastic media over which sewage or other wastewater flows downward and causes a layer of microbial slime (bio film) to grow, covering the bed of media. Formerly the trickling filter is thought to remove organic matter only by filtering it. However latter it was found that the major mechanism of organic removal in the trickling filter is microbial degradation rather than screening. This implies the name trickling filter does not stand for the trickling filter itself to trickle or filter.

Several factors are important in the function of trickling filters. Most important is to choose the correct filter media. The factors which are important in the selection of filter media are: void ratio, the volume that remains filled with air after the media has been filled into the filter housing divided by the total filter volume (Timmons and Losordo, 1997). Specific surface area is also important, the larger the filter media the larger will be the surface area where biofilm can grow per unit volume of filter medium, and increased removal of organics and nutrients per unit filter volume. The other advantage of larger specific surfacea area is smaller foot print. The filter media has also to be low weight for easy handling and it should facilitate homogenious flow of wastewater. The filter media must also have a reasonable price.

It is not surprising that trickling filters stimulate the growth of a complex mixture of microorganisms. A 2-arm rotary distributor provides a quick wastewater loading over the complete diameter of the circular trickling filter, followed by a rest period. Until the distributor arm passes over the same area again. Many engineers believed the bacteria responsible for metabolism of the contaminants in the wastewater needed additional time to complete metabolism. The wastewaters are quickly pulled by gravity through the trickling filter along the path of least resistance. The effluent is discharged from the bottom of the trickling filter in just a few seconds after being added at the surface. Even the high depth filter does not slow the wastewater flow significantly. Once the media has been wetted, surface tension will hold a thin liquid layer on the rock surface. The subsequent wastewater additions simply slide over the attached liquid as it provides the least resistance to flow. It is not surprising that the bacteria begin their growth and attachment in the cracks and crevices on the rock media surface and on the edges where rocks come into contact with each other.

As the bacteria growth extends out from the cracks and crevices along the rock media surfaces, the attached layer of water expands and extends beyond the bacteria surfaces. There is an exchange of liquid between the moving water wave and the attached water layer over the bacteria. The traveling water wave containing the waste contaminants are transferred into the fixed water layer that is kept attached to the surface of the bacteria. The travelling water wave pick up waste products from the attached liquid layer. This exchange processes results in a decreasing concentration of contaminants in the travelling wastewater as it flows through the trickling filter.

Aerobic conditions are maintained by splashing, diffusion, and either by forced air flowing through the bed or natural convection of air if the filter medium is porous and provides oxygen for the microorganisms growing as an attached bio film. During operation, the microorganism degrades organics matter, nitrify, denitrify etc depending on the operating condition. The biological slime grows in thickness as the organic matter abstracted from the flowing wastewater is synthesized into new cellular material.

The thickness of the aerobic layer is limited by the depth of penetration of oxygen into the microbial layer. The micro-organisms near the medium face enter the endogenous phase as the substrate is metabolized before it can reach the micro-organisms near the medium face as a result of increased thickness of the slime layer and lose their ability to cling to the media surface. The liquid then washes the slime off the medium and a new slime layer starts to grow. This phenomenon of losing the slime layer is called sloughing. The sloughed off film and treated wastewaters are collected by an under drainage which also allows circulation of air through filter. The collected liquid is passed to a settling tank used for solid-liquid separation.

Trickling filter is superior over the other more recent bio film reactors in that trickling filter need relatively low costs for construction and operation, a low excess sludge production as well as their being simple and robust. In addition trickling filters withstand peak loads and toxicity very well (Grady & Lim, 1980). Apart from the application of trickling filters for organic matter degradation by heterotrophic bacteria trickling filters have increasingly been used for tertiary nitrification of ammonium to nitrite and nitrate during the last decades. Trickling filters have also been applied for air stripping and gas cleaning, where the waste gases are fed through the filter. Some examples are odor control by H_2S removal (Gabriel & Deshusses, 2003). Toluene removal (Tseng *et al.*, 2001) and volatile organic contaminants in general (Vanhooren, 2001).

2.1 Operation

Distribution patter of the wastewater at the top of the trickling filter medium is very crucial operational factor. To exploit the maximum efficiency of the trickling filter the water should be distributed as evenly as possible. Beside this a uniform distribution of water helps in preventing channeling effect. The hydraulic loading rate should neither be very high nor very low otherwise in both cases poor bio film cover will result that reduces the capacity of the trickling filter.

In relation to the hydraulic load of the trickling filter there is one operational term which is called the dosing rate. The dosing rate is the depth of the wastewater applied on top of the trickling filter per unit time. For removal of organics high dosing rate increases efficiency as it is proved by Albertson & Davies (1984), among others. Greater wetting efficiency and removal of solids and filter larvae will occur as the dosing rate is increased. Low wetting rates can result in non uniform application of wastewater and unwetted areas that can attract snails and filter flies. “There is a marked decrease in efficiency of the trickling filter when there is no continuous wetting and supply of food by the wastewater”. The typical suggested hydraulic wetting rate range for BOD removal systems is 30 to 40 L/min•m^2 (0.75 to 1.0 gal/ min•ft^2). For nitrification systems, the range is 30 to 80 L/min•m^2 (0.75 to 2.0 gal/min•ft^2).

Observation of the trickling filter slimes under the microscope give rise to the visualization of the zoogleal masses of bacteria. No one was able to isolate the zoogleal bacteria in pure cultures until C.T Butterfield did in 1935. Working at the U.S public health service laboratory in Cincinnati, Ohio, Butterfield isolated Zoogleal ramigera in pure culture from activated sludge. In 1941 C.T Butter field and Elsie Wattie reported on the isolation of zoogleal forming bacteria isolated from trickling filters. The zoogleal forming bacteria isolated from trickling filters were similar to the zoogleal forming bacteria isolated from activated sludge (Ross E. McKinney, 2004).

2.2 The bio film

The total biology in the filter comprised of aerobic and facultative bacteria, fungi, algae, protozoa as well as higher animals depending on the condition it is exposed to. For standard trickling filter applications the two extremes are purely heterotrophic bio films and autotrophic bio films. The fungi (e.g.Pencillium, Geotrichum, Sporatichum and various yeast) on the other hand stabilize the waste but their role only becomes important at low pH in industrial wastes (Liu & Liptak, 2000). Under certain conditions their rapid growth can cause clogging of the filter. In the upper most part of the trickling filter where sunlight is available algae, such as Formidium, Chlorella and Ulothrix (Hawkes, 1963; Higgins & Burns, 1975), grow it can be assumed they have only a marginal effect on the trickling filter performance. In the outer portion of the film, organic pollutants ($C_aH_bO_cN_dP_cS_f$) are degraded by aerobic and facultative bacteria under aerobic conditions according to a biochemical reaction approximately expressed by the following equation:

$$C_aH_bO_cN_dP_eS_f + (4a + b - 2c - 3d + 5e + 6f)\ O_2$$
$$\rightarrow 4aCO_2 + (2b - 6d - 6e - 4f)\ H_2O + 4dNH_3 + 4ePO_4^{3-} + 4fSO_4^{2-} + (12e + 8f)\ H^+$$

Heterotrophic bacteria

At the interior of the bio trickling filter's bio film where oxygen is depleted the predominating bacteria are facultative bacteria. These bacteria decompose organic material in the wastewater along with aerobic bacteria and anaerobic bacteria. The most frequently encountered bacteria in the trickling filter are Acromobacter, Flavobacterium, Pseudomonas and Alcaligenes. Within the slime layer filamentous Sphaerotilus natans and Beggiatoa can be found (Metcalf & Eddy, 2003). Both aerobic degradation of organic matter and the use of other electron acceptors such as nitrate (denitrification), sulphate (sulphate reduction), hydrogen (methanogenesis) and other organic molecules (fermentation) can occur. The rapid formation of thick bio film (>10 mm) may happen therefore at high organic loadings if the aeration is sufficient and the hydraulic loading is moderate.

The performance of full-scale trickling filter is linked directly to physical characteristics (molecular size distribution) of biodegradable organic matter in the wastewater. Logan & Wagenseller (2000) have studied the influence of molecular size distribution of soluble organic matter on trickling filter performance. They found that smaller sized soluble organic matter molecules result in high rates of removal. At high BOD concentrations of dissolved organic matter, at close to neutral pH, high bulk water concentrations of oxygen and ambient temperatures substrate uptake rates up to more than 20 g BOD/m^2d can be achieved.

Autotrophic bacteria

In biological wastewater treatment system the oxidation of ammonium into nitrate (nitrification) occurs by aerobic bacteria which are carried out in two steps. First ammonium is oxidized into nitrite by bacteria commonly generalized to be of genus Nitrosomonas:

$$NH_4^+ + \frac{3}{2}O_2 \rightarrow NO_2^- + H_2O + 2H^+$$

Here there is reduction of the alkalinity of the effluent which is measured as equivalents of bicarbonates:

$$2H^+ + 2HCO_3^- \rightarrow 2CO_2 + 2H_2O$$

Further the nitrite oxidized to nitrate by genus Nitrobacter:

$$NO_2^- + \frac{1}{2}O_2 \rightarrow NO_3^-$$

However different researchers have found that nitrospira rather than nitrobacter are responsible for the conversion of nitrite to nitrate (Burell et al., 1998). The same is true in the findings of another researcher, Persson et al., (2002) and Lydmark et al. (in preparation). The latter researchers have detected no nitrobacter but nitrospira when they carry out studies on nitrifying trickling filters. Electron micro-graphs study carried out by Nevalainen et al., (1993) revealed that the morphologies of the ammonium oxidizers in their nitrifying bio films were similar to the species Nitrosomonas, Nitrosospira and Nitrosolobus. Studies using Fluorescent in situ hybridization (FISH) with 16S rRNA oligonucleotide probes in combination with con focal laser scanning microscopy (CLSM) for investigation of bio films in wastewater treatment have been carried out. And it has been confirmed that Nitrosomonas are probably the most common ammonium oxidizing bacteria in wastewater treatment (Wagner et al., 1995; Mobarry et al., 1996; Okabe et al., 1999), but Nitrosospira has also been detected (Schramm et al., 1998) and small numbers of Nitrosococcus (Lydmark et al., in preparation). According to the investigation done on the bio films along a nitrifying trickling filter, it is found that genus Nitrosomona were responsible for ammonia oxidation (Persson et al., 2002; Biesterfeld et al., 2001). Nitrosospira were not detected along the trickling filter height.

The characterization of bio films along the nitrifying trickling filter was also carried out using polymerase chain reaction (PCR) of 16S rRNA gene fragments combined with denaturating gradient gel electrophoresis (DGGE) by taking autotrophic bio film samples from different levels of a full-scale trickling filter and a higher number of ammonium oxidizing bacteria (AOB) probes (Rowan et al., 2003). And the result confirmed that several different Nitrosomonas strains coexisted. In addition variation in the dominant species type was observed between different levels of the trickling filter. This result was in agreement with the findings of the experiment done by combining real time PCR and FISH to quantify the AOB community at different depths of a full-scale tertiary nitrifying trickling filter (Lydmark et al., in preparation).

Variation in dominant species along the trickling filter height can be due to the different microbial environment that is favorable for some populations than for the others. For example, Nitrosomonas europea seemed to dominate at the top of the filer because they are known to like highly eutrophicated habitats, but disappear further down, where the ammonia concentrations are low most of the time. Where as in the lower levels of the filter, Nitrosomonas oligotropha was more abundant though dominant at all levels. Another interesting conclusion from the study by Lydmark et al. (in preparation) is that the AOB population change on a long term perspective. Earlier investigation made on the same plant reveals no detectable signals for Nitrosomonas europea (Persson et al., 2002).

A cryo-sectioned bio film from a nitrifying trickling filter that is analyzed with FISH illustrated the existancce of a close integration of AOB and nitrite oxidizing bacteria (NOB), as well as the abundance of other bacteria that may for example be heterotrophs that feed on degradation products and extracellular polymers. As obtained from the analysis result of a 9 mm long bio film slice, there exist bio film homogeneity with regard to the distribution of AOB and NOB on a higher level of aggregation.

Competition between autotrophs and heterotrophs

Several studies over the last few decades discovered that in an environment where oxygen and both ammonium and easily biodegradable organic matter are available in high concentration, the hetrotrophs will have a greater growth rates than the nitrifiers, the hetrotrophs are energetically favored. These two species compete for oxygen and space and in most cases the nitrifiers will eventually be out-spaced. The modeling of a one-dimensional bio film with respect to these species revealed the theoretical mechanics (Kissel et al., 1984; Wanner & Gujer, 1984) and be more stringently shown for any two species bio film (Wik & Breitholtz , 1996).

Down along a one-dimensional bio film cross section the biochemical reaction products are high in concentration where as the substrate concentration is decreasing this is mainly because the substrate transport into and out of the bio film occurred by means of diffusion (see Figure 2.1). As a result if the easily biodegradable organics are consumed already in the outer regions of the bio film, it is likely for the existence of the operating condition where the bacteria coexist as the bio film gets thicker. As a conclusion the coexistence of both the nitrifier and the hetrotrophs is possible when the ammonium concentration in the bulk water at the bio film surface is sufficiently high, the BOD concentration is not too high and the bio film is not too thin. Figure 2.2 illustrates theoretical steady state conditions where the AOB and heterotrophic bacteria coexist when the bulk water is saturated with oxygen (Wik & Breitholtz, 1996).

In practice the availability of the organic substance to the bacteria will determine the extent of hindering the nitrification process. Wastewater treatment plants having combined nitrification and degradation of organics need to have a sufficiently high sludge age (cell retention time in the system) because otherwise the nitrifiers will gradually be flushed out of the system. This is also one of the advantages with trickling filters and other fixed bio film reactors; since the nitrifiers are attached to the media they have long retention times even though the hydraulic retention time may be only 5–20 minutes.

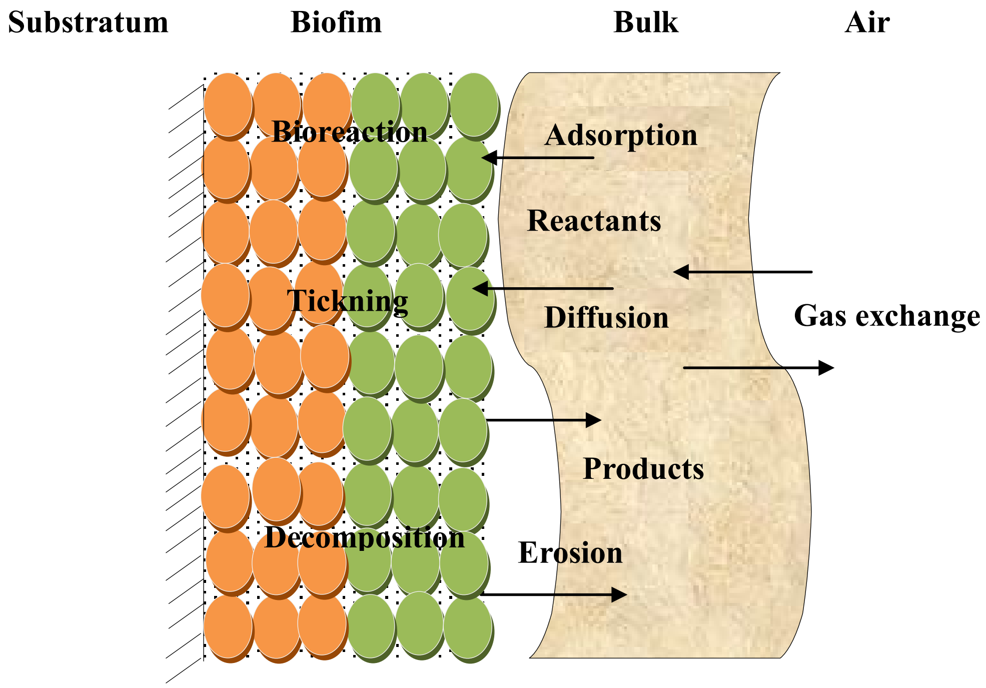

Figure 2.1 Essential trickling filter bio film concepts and phenomenon

A trickling filter that is fed with a fairly high ammonium concentration and BOD concentrations that are also fairly high exhibit unfavorable condition for the nitrifier at its top. However below some level in the filter the nitrifier will establish and may compete with the hetrotrophs because the BOD is low. These are not uncommon operating conditions for trickling filters used in wastewater treatment. Such a filter will have degradation of organic matter in the upper part that is gradually shifted to nitrification in the lower regions of the filter (Wik, 2003). An illustration of how the concentrations in the bulk water phase and in the bio film may look for a trickling filter with both organic degradation and nitrification is shown in Fig.2.2.

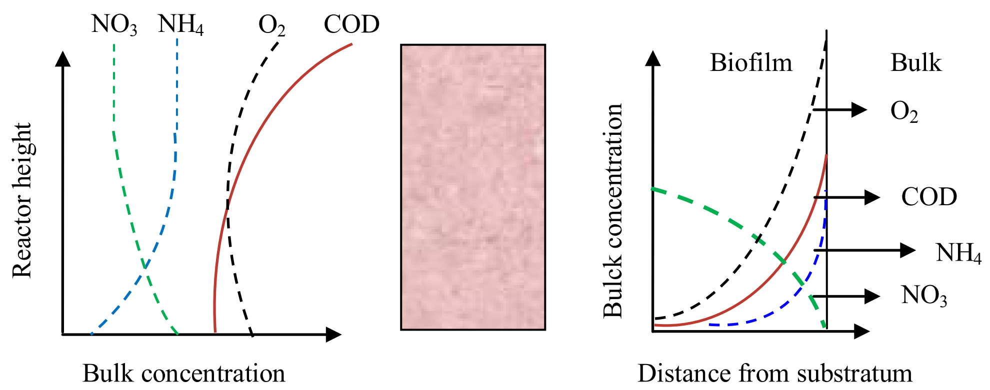

Figure 2.2 A conceptual transition of target chemical species and their concentration gradients in a trickling filter that involves both nitrification and organic matter degradation, assumed the biofilm (distance from substratum) and the bulk water (filter media height) aggregated to one dimension (adapted from Wik, 1999)

2.3 Fundamentals of trickling filter process modeling

Mathematical modeling is an important preliminary step for implementing wastewater treatment processes guiding system. Mathematical modeling remains the most efficient research method, even though it sometimes leads to abstract models that only approximately describe the structure of these processes. The mathematical models normally involve extreme simplification of the in situ phenomenon. However, the models help us to understand the basic properties of natural systems and search of mechanisms behind observed phenomena. To increase the knowledge about the dynamics of bio film reactors, modeling is an important tool, but the detailed models that arise are often very complex. Mathematically, they are systems of stiff nonlinear partial differential equations with a moving boundary (the bio film thickness). For optimization, controller design and for studies of large complex systems the mathematical models have to be neither detailed nor extremely accurate. However, the models have to be dynamic if, for example, they are to be used in controller design (Wik, 2003).

A continuously stirred bio film reactor unit

Commonly bio film reactors are modeled as a single continuously stirred bio film reactor (CSBR) or as a series of such. It should be stressed that in the following modeling if not otherwise pointed out, the bio film reactor is divided into n equal CSBRs. A gas phase compartment, a continuously stirred tank with bulk liquid, and a bio film compartment are the three compartments assumed in each compartments. Figure 2.3 illustrates the interaction between the three compartments. Every CSBR can be modeled separately of each other, i.e. bio film thickness, diffusion coefficients, bio film porosity, volume of bulk water, bio film area, reaction rate, kinetics and stoichiometry can be modeled in parallel for each CSBR. The methods and analysis will continue the same, but in outlook of the fact that they cannot be carried out in dimensionless time in a clear-cut manner, indexing and expressions become rather bulky.

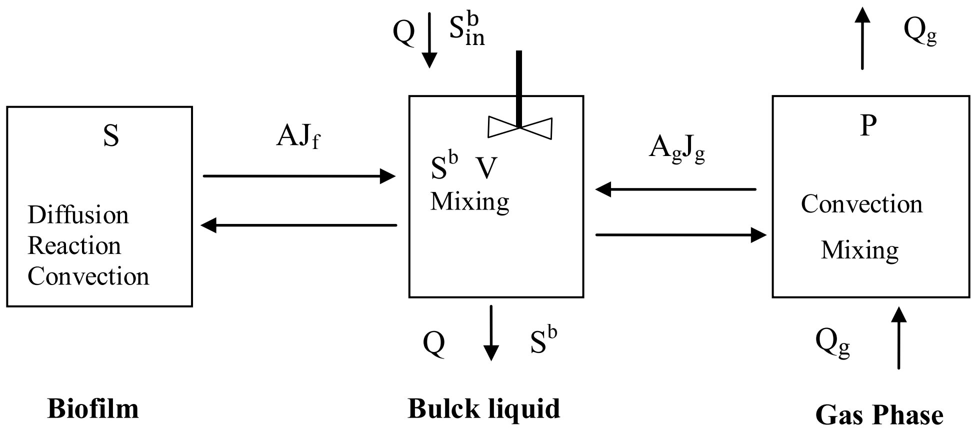

Figure 2.3 A continuously stirred bio film reactor unit with no reaction in the bulk

Using the variables in the figure, mass balances for dissolved substances over the bulk give

$$V\frac{d}{dt}S_i^b = Q\left(S_{in,i}^b - S_i^b\right) + AJ_{f,i} + A_gJ_{g,i}\,,\; i = 1,2,\ldots,n_l \tag{1}$$

Where V is the bulk water volume (m^3), S^b is the substrate concentration (g/m^3) in the bulk, A the area of the bio film (m^2), A_g the contact area between gas phase (air) and bulk water, J_g the flux (g/m^2d) from gas to bulk, J_f the flux from bio film to bulk and n_l the number of substrates considered in the model. The flux out of the bio film is equal to the flux at the bio film surface, i.e.

$$J_{f,i} = -D_i\left[\frac{dS_i}{dx}\right]_{x=L} \tag{2}$$

Where S is the concentration in the bio film the distance (m) from the substratum, Di the diffusion coefficient for substrate i, and L the bio film thickness. Here it is assumed a one-dimensional planar bio film and transport of substrates mainly by Fickian diffusion. The substrate concentration derivative has to be determined by combining (1) and (2) with a model of the bio film. Such models may have very different complexity depending on the purpose of the modeling. For cylindrical and spherical bio films the expressions are similar (Wanner & Reichert, 1996; Wik, 1999a).

In aerobic bio film systems, the reaction rates usually depend on the oxygen concentration. In well aerated trickling filters with moderate oxygen consumptions, the bulk may be assumed saturated with oxygen. At higher respiration rates empirical approximations of the oxygen flux can be made, which is discussed in the next section. The total flux of a gaseous substance, AgJg,i, depends on the contact area between the bulk and the gas phase, the partial pressure pi, the solubility, the bulk concentration, the mixing in the gas phase et cetera.

2.3.1 Reactor Modeling

Hydraulic Modeling

By combining CSBR units in parallel and in series, different hydraulic behavior can be modeled in the same way as continuously stirred tank reactors (CSTRs) are combined in traditional chemical reactor engineering and design (Froment & Bischoff, 1979).

In trickling filter, though, the mixing cannot be idealized and combinations of CSBRs in parallel and in series may better approximate the hydraulics. Ideally, the flow through a trickling filter and some other bio filters should be a plug flow, i.e. no vertical mixing but complete horizontal mixing. A mass balance over a reactor segment d_z in such a reactor gives

$$V\frac{d}{dt}S_i^b = \mathrm{q}\frac{\mathrm{d}}{\mathrm{dz}}\mathrm{S}_\mathrm{i}^\mathrm{b} + \mathrm{aJ}_{\mathrm{f,i}} + \mathrm{a_g J_{g,i}},\ \mathrm{i} = 1,2,\dots,\mathrm{n_l} \quad (3)$$

where z is the level in the filter. If the reactor is uniform, the specific bulk volume v, the hydraulic load q, the specific bio film surface area a, and the specific surface area of the gas–liquid interface a_g are defined as: $\mathrm{v} = \mathrm{V}/(\mathrm{A_r h})$, $\mathrm{q} = \mathrm{Q}/\mathrm{A_r}$, $\mathrm{a} = \mathrm{A}/(\mathrm{A_r h})$ and $\mathrm{a_g} = \mathrm{A_g}/(\mathrm{A_r h})$

where A is the total bio film area (m^2) in the filter, V the total bulk water volume (m^3), A_g the total gas-bulk interfacial area (m^2), A_r the cross-sectional area (m^2) of the reactor and h the reactor length (m) or the reactor height. Equation (3) corresponds to an infinite number of CSBRs in series. Instead of solving this partial differential equation, the plug flow reactor may therefore be approximated by a large number of CSBRs in series. It is therefore natural to model a trickling filter as N CSBRs in series, where N can be seen as a model design parameter: the higher the N the closer to plug flow. An alternative hydraulic model is to model the bulk concentration as a dispersed plug flow reactor (Seguret et al., 2000).

2.3.2 Gas mass transfer

Oxygen transfer in TFs is an important design criterion since BOD removal in excess of oxygen availability to the biofilm can create anaerobic conditions and cause odors (Logan et al., 1989b). If the oxygen supply is correctly modeled, the equations for all CSBRs have to be solved simultaneously. It is, therefore, numerically advantageous if the oxygen flux from the gas phase to the liquid, or the oxygen bulk concentration, can be approximated such that the causality of the bulk flow can be used. Then we can solve the equations for one CSBR at a time (from top to bottom), which makes a huge difference in computational demand. A schematic description of such a trickling filter model is shown in Fig.2.4.

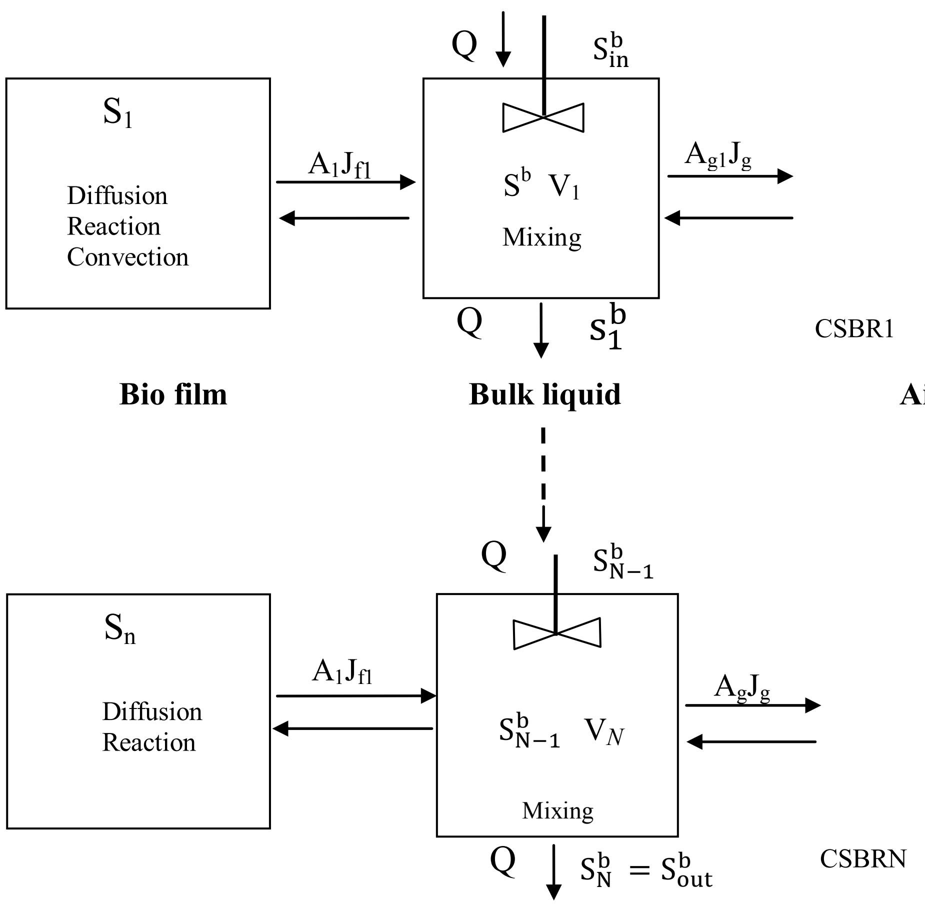

Figure 2.4 Structure of a causal trickling filter model with N CSBRs in series

At low respiration rates and proper ventilation it is natural to assume that the bulk is saturated with oxygen. At high respiration rates, though, the oxygen has to dissolve in the bulk liquid at a higher rate. The driving force has to be higher then, which implies reduced oxygen bulk concentrations. To estimate the oxygen mass transfer we may use a standard expression (Wik, 2003).

$$j_{g,O_2}(Z) = K_{L,O_2}\left(S_{O_2}^{sat}(T) - S_{O_2}^{b}(Z)\right), \tag{4}$$

where T is the water temperature. By sampling along the axis of the filter and using stoichiometry the oxygen mass transfer coefficient can be estimated.

2.3.3 Trickling filter fast dynamics

To optimize operation and guarantee stable control systems the fast dynamics often have to be taken into consideration in the daily operation of a trickling filter and other bio film reactors. When the substrate load varies quickly, the fast dynamics also play an important role (Rittman, 1985). Further, since physically based models of the fast dynamics are in many ways simplifications of more complex models of the slow dynamics, important model parameters are the same (Kissel et al., 1984; Gujer & Wanner,1990; Wik & Breitholtz,1998; Wik, 1999a). That implies the parameters identified from experimental data using the fast dynamic models, can also be used for acquiring information about the slow dynamics as well.

Wik et al. (1998, 1999a, b, c, in prep.) considers the different aspects of modeling the fast dynamics of bio film reactors more comprehensively. Here, we only derive model equations for a simple CSBR model in which the assumptions are; fickian diffusion of substrates transports in the bio film, which is assumed to be homogeneous and planar. Further, the bulk concentration and the concentration at the bio film surface are identical, and the reaction rate is linearly related to the substrate considered. If the mass transfer between the bulk and the gas phase is assumed to be zero, Equation (1) can be written in non-dimensional form as (Wik, 2003)

$$\tau \frac{d}{d\tilde{t}} S^b = S_{in}^b - S^b - \gamma \left[\frac{ds}{d\tilde{x}}\right]_{\tilde{x}=1} \tag{5}$$

where $\tau = \frac{V}{Q}\lambda$, $\gamma = \frac{AD}{QL}$, $\lambda = \frac{D}{L^2\varepsilon}$,

ε is the bio film porosity (m^3/m^3), τ is time constant (non-dimensional or d), γ is non-dimensional coefficient for substrate flux into biofilm and λ is time scaling coefficient, time is scaled as $\tilde{t} = \lambda t$ and the distance to the substratum is scaled as $\tilde{x} = x/L$. Now, if the reaction rate in the bio film is assumed to depend linearly on the substrate concentration the equation describing the concentration in the bio film is (Wik, 2003)

$$\frac{ds}{d\tilde{t}} = \frac{d^2S}{d\tilde{X}^2} - kS - \mu, \qquad 0 < \tilde{x} < 1 \tag{6}$$

where μ is the zero order rate coefficient and k the first order rate coefficient. When the reaction rate (g/m^{3}d) is described by Monod expression, i.e.

$$r = r_o \frac{S}{S+K} \quad . \tag{7}$$

When the substrate concentration is very high i.e. S ≫ K, the rate considered to be zero order with $r = r_o$, which gives $\mu = r_o L^2/D$ and k = 0. At low substrate concentration the rate considered to be of first order dependence $r = r_o s/K$, which gives μ = 0 and k= $r_o L^2/KD$. It is possible to linearize the Monod expression (7) around an arbitrary substrate concentration $\bar{s}$ if we want to focus on slight change near to an operating concentration. The point now is that equations (5) and (6) are linear and we may therefore derive a transfer function that describes how changes in the effluent bulk concentration S^b depend on arbitrary changes in influent bulk concentration s_{in}^b (Wik, 2003).

$$G(S) = \frac{\Delta S^b(S)}{\Delta S_{in}^b(S)} = \frac{1}{1+\tau S+\gamma\sqrt{S+k}\ \tanh\sqrt{S+k}} \tag{8}$$

Where G is transfer function, transfer function matrix, $\Delta s^b(s)$ and $\Delta s_{in}^b(s)$ are the Laplace transforms of $\Delta s^b(t) = s^b(t) - s^{-b}$ and $\Delta s_{in}^b(t) = s_{in}^b(t) - s_{in}^{-b}$ where s^{-b} and s_{in}^{-b} are the steady-state concentrations in a (constant) operating point. This transfer function can be applied in any frequency based controller design. The transfer function of a trickling filter modeled as N CSBRs in series can be represented as simply the product of N transfer functions given by (8). This transfer function can be extended to several substrates and it may also be used for describing the behavior when there are variations in the flow Q as well as in the influent concentrations (Wik, 1999c). A very interesting feature of the transfer function (8) is that it can be closely approximated by a low order rational transfer function on the form (Wik, 2003)

$$\widehat{G}(s) = G(O) - K_1 - k_2 - \cdots - k_m + \frac{k_1}{1+sT_1} + \frac{k_2}{1+sT_2} + \cdots + \frac{k_m}{1+sT_m} \tag{9}$$

or if a strictly proper transfer function is required

$$\widehat{G}(s) = \frac{G(O)}{K_1 + K_2 + \cdots + K_m} \left(\frac{k_1}{1+sT_1} + \frac{k_2}{1+sT_2} + \cdots + \frac{k_m}{1+sT_m}\right), \tag{10}$$

where $G(0)= 1/\left(1+\sqrt[r]{k}\tan\sqrt[h]{k}\right)$ describes how a stationary change in the influent concentration causes a change in effluent concentration, i.e. $\Delta s^{-b} = G(0)\Delta s_{in}^{-b}$ scaling back to original time is achieved by dividing s in the transfer functions by the time-scaling factor λ. The higher m is the closer the approximations are to (8).

2.3.4 Residence time distributions

Determination of model parameters from residence time distributions (RTDs) is a standard procedure in chemical reactor analysis and design. Typically, a trace substance is dissolved in a small volume and added to the influent to the reactor. Assuming that the duration of the addition can be ignored, the effluent concentration of the trace substance is related to the unit impulse response according to (Wik, 2003)

$$S_{out}^{b}(t) = \frac{m_\delta}{Q} g(t) \tag{11}$$

where m_δ the mass of the added trace substance and t is the time after the addition. The unit impulse response of a trickling filter model with N CSBRs in series is the inverse Laplace transforms of the product of all CSBR transfer functions. Two common methods of parameter estimation from RTDs are least square fitting of simulated responses to measurement data and determination using the measured moments around the mean residence time. The latter method is usually easier to implement and also more rapid, but may give poor results if the measured concentrations do not agree well with the ones predicted by the model (Wik, 2003). From an investigation of RTDs for cascaded CSBRs (Wik, 1999a) it is evident that the values of ε, L, A and N have significant effects on the responses, while the diffusion coefficient only has a small effect on the impulse response. It is illustrated that the methods applied to the case of N cascaded identical CSBRs. The bio film area in each CSBR is then A/N, the bulk water volume is V/N and the bio film porosity ε and thickness L are the same in all CSBRs.

Larger values of ε, L and A give pronounced tailing and slower responses, which can be attributed to an increased hold-up of substance in the bio film liquid volume ε LA. Changes in these parameters affect the shape of the responses differently. Changes in A and ε reshapes the entire responses, while changes in L mainly affect the tails of the responses.

2.3.5 Trickling filter slow dynamics

The origin of the dynamics in the bio film reactors can be divided as change in biology, transient of the dissolved substance in the liquid phase of the bio film and transients in the bulk liquid. Usually it takes days for the bacterial composition in the bio film to change, while it only takes a few minutes for the concentrations in the bio film to settle after changes in the bulk concentration. The hydraulic modes are also fast, where the transients settled in less than half an hour. When the long term effects of the conditions for a bio film reactor are to be studied, the dissolved components can therefore be assumed to be in a steady state. Microbial processes, such as the growth and decay of the bacteria, will govern the reactor behavior (Wik, 2003).

It is possible to model the slow dynamics in varying degree of complexity. The most complex models combine a hydraulic model, such as CSBRs in series, with multispecies models where the spatial distribution of the bacteria in the bio film is included (Kissel et al., 1984; Wanner & Gujer, 1986; Wik, 1994). The microbial transformations in the bio film model are illustrated in figure 2.5. An increased sludge age is associated with a decreased sludge production. This phenomenon is generally interpreted as a result of endogenous respiration processes. In the model cell lysis (or decay) is incorporated. The lysis is modelled such that it leads to generation of particulate substrate, which by a hydrolysis process is converted into soluble substrate. The substrate is then converted to biomass again by growth processes. The work of Wanner & Gujer (1984, 1986) and Wanner & Reichert (1996) has evolved into the software AQUASIM, while the numerical ideas of Kissel et al. (1984) was carried on by Wik (1999a) and can now be used in a Matlab environment. Less complex models that are still based on the slow dynamics have been described by for example Rittmann (1989), Rittmann & Manem (1992) and Vanhooren (2001).

In a trickling filter the death of bacteria is a complex phenomenon. Death may result from substrate depletion, toxic substances and invasion by viruses, for example. Inability to form colonies and grow does not imply death. Cells can be merely injured and recover given the right growth conditions. Like most living species, the cells do not die instantly from starvation. Instead, they remobilize inner resources and stop growing when the growth conditions become too unfavorable. This transformation is sometimes called exogenous dormancy. If they starve for too long, they will eventually die, but, if the conditions become appropriate before this occurs, they may quickly reactivate. In Figure 2.6 the described biological transformations are illustrated. A process that can be significant is the maintenance process, which is not a true transformation and therefore not shown in the figure. The maintenance requires energy and reflects a diversion of substrate away from growth. Consequently, the maintenance decreases the observed yield of cells from substrate.

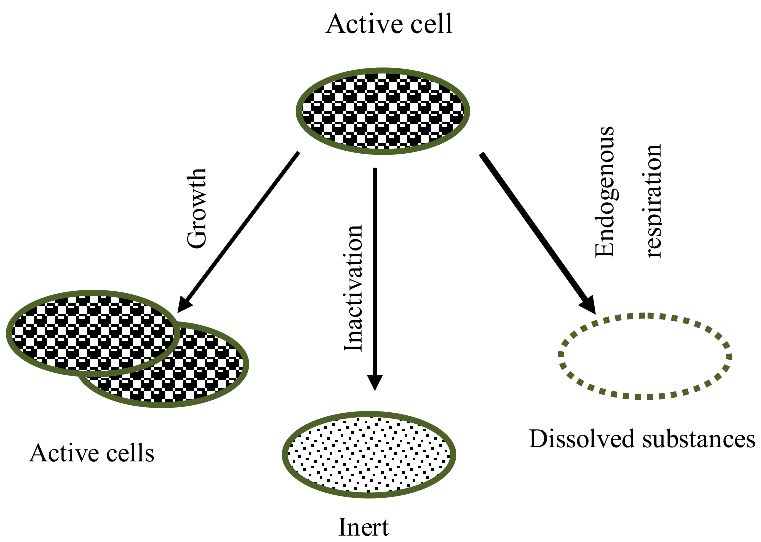

Figure 2.5 Traditional multi species model for microbial dynamics in a trickling filter

The best way of including dormancy in bio film models for bio film reactors requires more knowledge about the starvation processes of the modeled species. However, a few ideas can be drawn from experimental studies on other bacteria. The inactivation, i.e. when the bacteria go from the active reproductive state into the dormant state, begins almost instantly after substrate limitation. In the inactivation process, the cells may shrink and increase in numbers (Amy et al., 1983), which gives the cells a higher area to volume ratio and, consequently, an improved ability for substrate uptake. When inactivated, the endogenous respiration may decrease substantially (Novitsky & Morita, 1977; Nyström 1989 and references there). The activation process may occur within a few hours after the growth conditions have become appropriate (Kjelleberg et al., 1982; Amy et al., 1983; Horn & Hempel, 1997b). If the cells have decreased in size when inactivated, they regain a larger size again.

Using the transformations in figure 2.6 an extension of the bio film model that includes dormant cells was made (Wik 1999a, 2000). It was then found that to what extent the capacity is increased by the simulated strategies depends particularly on the specific death rate and the true activity of the nitrifiers, which needs to be further, studied. The possible use of micro autoradiography in combination with FISH, PCR, DGGE and bio film modeling appears to be a promising way to resolve these questions.

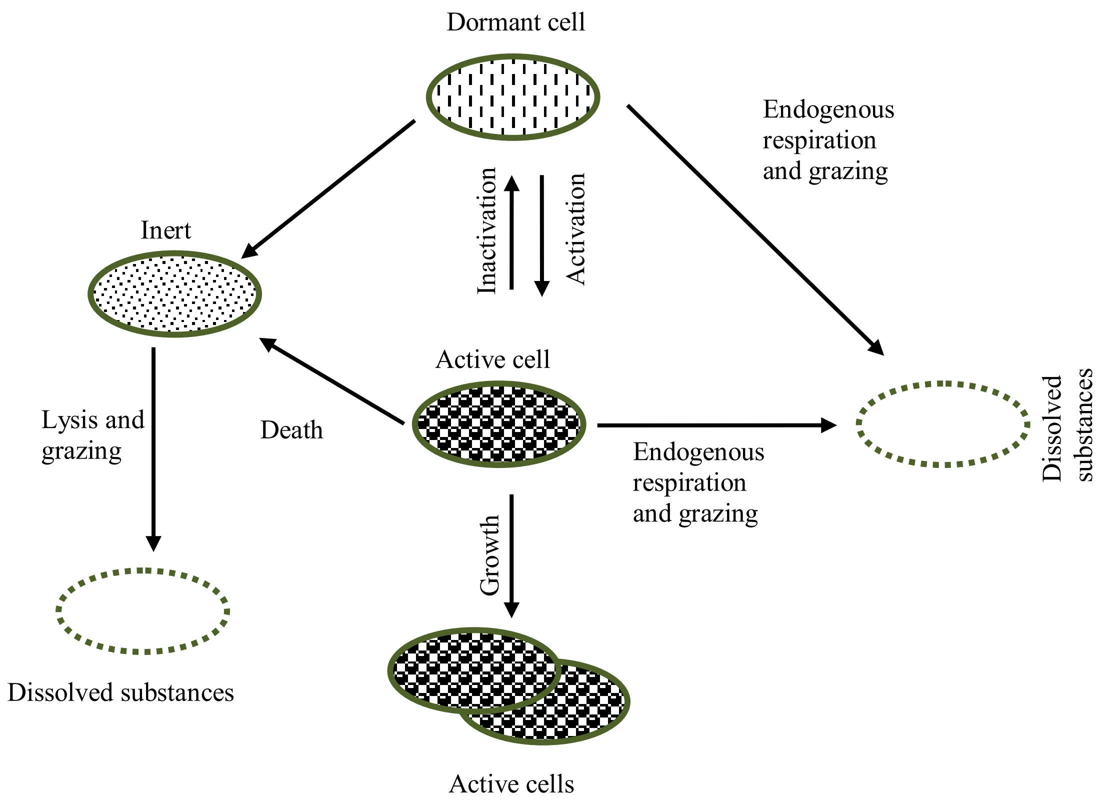

Figure 2.6 Illustration of microbial transformations in an extended bio film model

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3 A pilot scale trickling filter with pebbles gravel as media and its performance to remove chemical oxygen demand from synthetic brewery wastewater

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Abstract

Evaluating the performance of a bio trickling filter for the treatment of brewery wastewater was the aim of this study. A pilot scale trickling filter filled with gravel was used as the experimental bio filter. Pilot scale plant experiments were made to evaluate the performance of the trickling filter aerobic and anaerobic bio film systems for removal of chemical oxygen demand (COD) and nutrients from synthetic brewery wastewater. Performance evaluation data of the trickling filter were generated under different experimental conditions. The trickling filter had an average efficiency of 86.81 ± 6.95 % as the hydraulic loading rate increased from 4.0 to 6.4 $m^3/(m^2{\cdot}d)$. Various COD concentrations were used to adjust organic loading rates from 1.5 to 4.5 kg COD/($m^3{\cdot}d$). An average COD removal efficiency of (85.10±6.40) % was achieved in all wastewater concentrations at a hydraulic loading of 6.4 ($m^3/(m^2{\cdot}d)$). The results lead to a design organic load of 1.5 kg COD/($m^3{\cdot}d$) to reach an effluent COD in the range of 50–120 mg/L. As can be concluded from the results of this study, organic substances in brewery wastewater can be handled in a cost-effective and environmentally friendly manner using the gravel-filled trickling filter.

Key words: Biodegradation, Pilot scale trickling filter, Aerobic treatment, Brewery wastewater, Chemical oxygen demand (COD), Trickling filter performance

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3.1 Introduction

In the last 20 years environmental awareness of the brewing industry has grown significantly leading to increased investments in environmental protection measures (Driessen & Vereijken, 2003). Different countries place different systems based on the country's as well as the breweries' specific condition in order to minimize the pollution contribution from this sector of industry. Effluent treatment requirement of breweries based on their discharge is one of the different systems (World Bank, 1997). Substantial improvements has been made in the past, however it has been estimated that approximately 3 to 10 L of waste effluent is generated per liter of beer produced in breweries and the wastewater to beer ratio is only 1.2 m^3/m^3 to 2 m^3/m^3 less with the remaining water disposed off with by products and lost by evaporation (Driessen & Vereijken, 2003).

"Un treated brewing effluent pose a significant treat to surface and ground water qualities". The main brewery effluent sources include losses during bottle filling, cleaning (of returned bottles, fermentation and conditioning tank, vat and floors) and draining tank bottoms (Yu, H.and Gu, G., 1996; Driessen et al., 2003).The composition of brewing effluent can fluctuate significantly as it depends on the process (Table 3.1). Untreated brewery effluent typically contains suspended solids (TSS) (200 – 1000 mg/l), biochemical oxygen demand (BOD) (1,200 – 3,600 mg/l), chemical oxygen demand (COD) (2,000 – 6,000 mg/l), nitrogen (N) (25 – 80 mg/l), phosphorus (P) (10 – 50 mg/l), temperature in the range of 18°C−40°C, and a pH between 3−12 (Driessen et al., 2003; IFC, 2007). The high organic content of brewery effluent classifies it as a very high strength waste (Schwartz, H.G., Jr., and Jones, M.R.H, 1971; Kanagachandran, K., and Jayaratne, R., 2006), meaning that the brewery effluent cannot simply be discharged into sewers or water courses. Direct discharge can bring about a rapid deterioration of the physical, chemical, and biological qualities of the receiving water bodies (The Breweries of Europe, 2002; Parawira et al., 2005; Al-Rekabil et al., 2007).

The decomposition of organic matter depletes the amount of dissolved oxygen in the water that is vital for aquatic life. Release of nitrogenous and phosphorous compounds in the wastewater also stimulates aquatic plant growth contributing to eutrophication of water bodies. Furthermore turbidity and color reduces the penetration of light, which, in turn, affects photosynthesis, thereby affecting the primary link in the food chain. The removal of organic compounds from the wastewater is important to avoid anaerobic conditions in the receiving waters. Nutrients like nitrogen (N) and phosphorous (P) should also be removed to avoid algal blooms that disturb the ecosystem of the receiving waters (Driessen et al., 2003).

"The implimentaion of trickling filters, for which the functional unit is the biofilm for untreated industrial wastewater discharge may turn into a convenient solution, especially for small urban communities without significant investments on capital and operation costs. This is true in most developing countries, where the discharge of untreated wastewater streams is still a common practice, including domestic sewage." A biofilm can be characterized as an organic matrix consisting of a complex community of bacteria, algae, fungi and protozoa embedded in organic polymers. In fixed biofilm reactors the biofilm is attached to substrata that, generally, are impermeable. Substrates diffuse from the bulk liquid into the biofilm where the bacteria carry out the desired transformations of the substrates. Reactors of this kind have attained increased attention during the last three decades, particularly in drinking water and wastewater treatment, due to the ability to withhold bacterial populations having lowgrowth rates, and new materials that give high specific capacities (Chaudhry and S.A. Beg, 1998; Wik, 1999).

In addition to this the attached growth biofilm systems rendered several advantages over the suspended growth biomass systems. The specific advantages vary with the type of biofilm system and reactor configuration. In general, a biofilm system offers the following advantages (Tchobanoglous, 1995):

- High biomass packing density and reactor compactness due to a large specific surface area (smaller foot print)
- Short contact periods and co-habitation of aerobic and anoxic micro-organisms within the same ecosystem
- Reduced sludge bulking and better sludge thickening qualities
- Lower sensitivity and better recovery from shock loadings
- Low energy requirements and more economy in operation and maintenance
- Low sludge production and superior process control
- Simple in operation and maintenance.

The Electrical Power Research Institute (EPRI) has studied power usage for trickling filter wastewater treatment plants and activated sludge wastewater treatment plants. They found that trickling filter plants consume approximately 70% of the electricity consumed by activated sludge plants (EPRI, 2002). Over the years, the treatment of wastewater using biofilm technologies has been established to be an efficient and proven technology with relatively stable end-products. They offer an ideal alternative, mainly as a secondary or tertiary biological treatment unit for the simultaneous removal of organic substances, nitrogen and other nutrients in municipal wastewater (Müller et al., 1980, Masuda et al., 1990, Boller et al., 1990).

Therefore in this study, the detailed investigation aimed at analyzing the performance of gravel-filled, naturally aerated trickling filter on brewing industry wastewater. And to demonstrate the use of a trickling filter as an alternative biological process over conventional activated sludge process with respect to cost.

Table 3.1 Characteristics of some industrial brewery and the local brewery waste water

Parameter/benchmark per unit	Brewery effluent composition		
	Rostocker*	Typical •	St.George ••
COD [mg/L]	1600-9000	2000-6000	1860-3880
BOD [mg/L]	1200-8000	1200-3000	484-636
TSS** [mg/L]		200-1000	793-5048
TS** [mg/L]			1554-7548
T [°C]	18-35	18-20	23-34
pH [-]	4-12	4.5-12	6.0-11
Nitrogen [mg/L]	30-120	25-80	44-78
Phosphorous [mg/L]	10-45	10-50	

*The local brewery

** TSS, TS: total suspended solids, total solids

• (Driessen & Vereijken, 2003)

•• (Shumete Y.& Leta S., 2008 in preparation)

3.2 Materials and methods

3.2.1 Pilot scale trickling filter

The schematic diagram and photo of the pilot scale trickling filter is shown in Figs. 3.1 and 3.2 respectively. The pilot scale trickling filter consisted of a plexiglas tube with an inner diameter of 40 cm and a total height of 180 cm. Sampling ports are located at fixed intervals of 260 mm along the height of the bio filter. At the top of the filter, a fixed flow distributor was installed to facilitate a uniform distribution of the wastewater fed to the filter's free surface and a perforated tube connected to the fed pump and sprays the fed water coiled and placed over it. Wastewater from a storage tank (300 L in volume) is being homogenized and introduced at the top of the reactor. Also, a secondary clarifier was installed to collect and settle the effluent from the filter's draining system. The supernatant from the secondary clarifier was collected in a 10 liter container for recirculation via a recirculation pump.

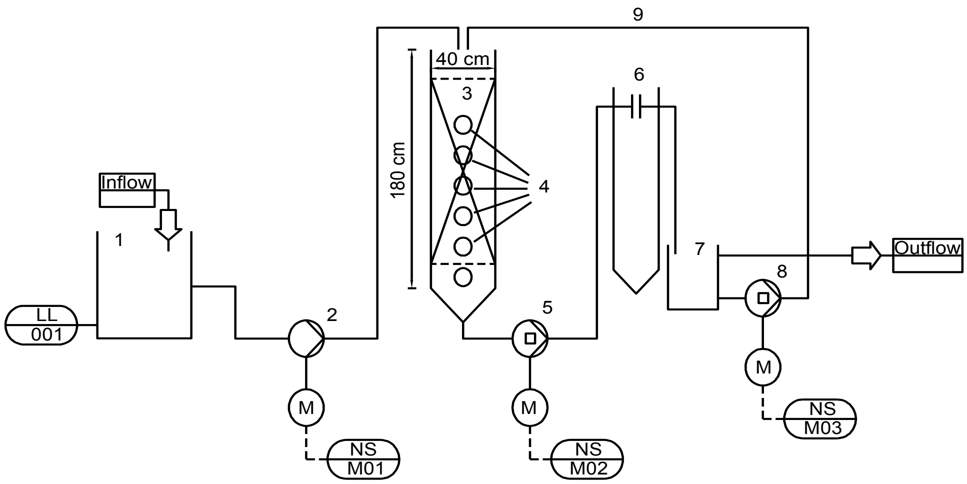


Figure 3.1 Flow scheme of the experimental setup-pilot scale trickling filter

1: wastewater reservoir; 2: fed pump; 3: trickling filter; 4: sampling ports; 5: draining pump; 6: secondary clarifier; 7: clarified water; 8: recirculation pump; 9: recirculation

Figure 3.2 The pilot scale trickling filter

3.2.2 Filter medium

In trickling filters the ideal filter medium is a material that has a high surface area per unit volume, is low in cost, has a high durability, and does not readily clog. The choice of filter media is more often governed by the material locally available which may include field stone, gravel, broken stone, blast furnace slag and anthracite stones. Therefore in this study the supporting media used was pebble gravel purchased from a gravel producer with very small price. The diameter of the filter stones, as obtained from the specification of the gravel producers, was in the range of 16-64 mm. At the bottom of the trickling filter larger stones (80-100mm) was placed that is used as a support. Then about 90 % of the 32-64 mm and 5 % of the 16-32 mm size groups were placed at the middle and top layers of the trickling filter respectively. Figure 3.3 below shows the photo of the three size group of trickling filter media. Under operating conditions nearly 2/3 of this can be assumed to be biologically active (ATV-DVWK-A 281 E, 2001). The filter which is packed in such a manner had a specific surface area and void ratio of 72 m^2m^{-3} and 45 %, respectively. The total depth of the filter media was 160 cm including the support stones at the base of the reactor.

Since the placement of the filter media is of paramount importance to the efficiency of the percolating filter, it was carried out with particular care and under proper control. Filter media were placed without prolonged intermediate storage and was packed in such a way that the largest possible intervening spaces resulted in the trickling filter. In order to ensure that as little abrasion occurs and a separation is avoided appropriate handling equipment was used and the drop height not exceeding 500 mm.The abraded matter are removed from the trickling filter by washing with sufficiently large amount of water in order to avoid zones which are impermeable to water and air. The packed filter media were washed with a sufficiently large volume of tap water and checked for flooding, flow rates, and hydraulic loading rates of the operation (flow rates up to 556 ml/min or hydraulic loadings up to 637 ml/ ($cm^2 \cdot d$)).

Figure 3.3 River stone medium: A (16-32mm); B (32-64mm) and C (80-100mm, Support)

3.2.3 Trickling filter operation

A Synthetic brewery wastewater was prepared using ammonium sulphate, disodium hydrogen phosphate, ethanol, malt extract, maltose, peptone, sodium hydrogen phosphate and yeast extract as ingredients (Boeije, 1996). A concentrated substrate was used and later diluted with tap water to prepare the feed synthetic wastewater. Mode of operation of the trickling filter was intermittent flow with recirculation during all trickling filter operations. The first operation was the seeding of the trickling filter with real wastewater. Then followed the acclimation of the trickling filter using first real wastewater, then a mixture of both real and synthetic wastewater, and finally only with synthetic water until the system reaches its acclimation period. COD reduction was monitored to evaluate the growth of the microbial population. Atainment of the acclimation period was realized when the COD removal for three consecutive days were nearly the same.

The next operation was the investigation of the effect of hydraulic and organic loading rates on the performance of the system. During this operation of the trickling filter, the performance was evaluated at four different hydraulic and organic loading rates. At each hydraulic and organic loading rate condition, the trickling filter was operated for about five consecutive days to ensure repetitiveness of the result. The performance of the trickling filter was also evaluated for different combinations of daily influent flow and influent concentrations. Initial influent sample analysis was done at the beginning of each run in all cases and final effluent sample analysis was done three times a week to investigate the effect of hydraulic and organic loading rates unless otherwise indicated and on a daily basis to investigate different combinations of daily influent flow and influent concentrations. The performance of the trickling filter for the temperature range of 20 – 40 °C and pH range of 3–12 was also investigated. The other investigation during this operation of the trickling filter was COD profile along the trickling filter height.

3.2.4 Ventilation of the trickling filter

Air is driven through the trickling filter by vertical pressure differences developed by thermal buoyancy. The warm air inside the bioreactor is less dense than cooler air outside, and thus will try to escape from openings high up in the trickling filter column, cooler denser air will enter openings lower down. The process will continue if the air entering the bioreactor is continuously heated, typically by casual or solar gains (Linden, P. F., 1999). Due to the elongated nature of the trickling filter column a greater buoyancy force that governs the movement of air could occur in this trickling filter. Large enough ventilation ports are provided at the bottom of the filter and the medium has enough void area. Natural ventilation as an alternative to mechanical ventilation has several benefits: low running cost, zero energy consumption, low maintenance and probably lower initial cost of the trickling filter.

Passive devices for ventilation of the trickling filter were present in the form of vent stacks on the trickling filter periphery, extensions of under drains through trickling filter sidewalls (the under drain is angled to admit air). The construction of the trickling filter also involves ventilating manholes, louvers on the sidewall of the tower near the under drain. During operation discharge of trickling effluent to outside in an open channel or partially filled pipes was practiced to inhance further aeration. Also the under drain was kept always half filled.

3.2.5 Instrumentation and analytical methods

Wastewater samples for determination of the physicochemical parameters were collected in 10 liters polyethylene bottles by direct immersion of the containers. Sample was taken from the point, a pond like structure, where discharges from different brewing sections (brew house, fermentation, filtration, and packaging) as well as from cleaning activities mix together before they leave the brewery.

The raw brewery wastewater which is obtained from the local brewery was first subjected to roughing screens, PH adjustment using NaOH and HNO_3, and defoaming by the brewing company. The wastewater samples were then analyzed for COD, nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), ammonium nitrogen (NH_4-N), total nitrogen (TN), phosphate (PO_4^{3-}), and total phosphorus (TP) colorimetrically. Prior to analysis, the soluble fraction of the wastewater samples was obtained by filtering with a syringe filter of 25 mm diameter (W/0.45 µm cellulose) for analyses of NO_3-N, NO_2-N, NH_4-N, and (PO_4^{3-}). An unfiltered wastewater sample was used for analyses of COD, TN and TP. The basic principles governing the analytical techniques in the present investigation is discussed here under

The chemical oxygen demand (COD) test is commonly used to indirectly measure the amount of organic compounds in water. Most applications of COD determine the amount of organic pollutants found in surface water (e.g. lakes and rivers) or wastewater, making COD a useful measure of water quality. It is expressed in milligrams per liter (mg/L) also referred to as ppm (parts per million), which indicates the mass of oxygen consumed per liter of solution.

The basis for the COD test is that nearly all organic compounds can be fully oxidized to carbon dioxide with a strong oxidizing agent under acidic conditions. The amount of oxygen required oxidizing an organic compound to carbon dioxide, ammonia, and water is given by:

$$C_nH_aO_bN_c + \left(n + \frac{a}{4} - \frac{b}{2} - \frac{3}{4}C\right)O_2 \rightarrow nCO_2 + \left(\frac{a}{2} - \frac{3}{2}C\right)H_2O + CNH_3$$

Biochemical oxygen demand (BOD) is the amount of dissolved oxygen needed by aerobic biological organisms in a body of water to break down organic material present in a given water sample at certain temperature over a specific time period. It is widely used as an indication of the organic quality of water (Clair N. Sawyer *et al.*, 2003). During the determination of BOD the sample is kept in a sealed container fitted with a pressure sensor. A substance that absorbs carbon dioxide, sodium hydroxide in this investigation is added in the container above the sample level. The sample is stored for 5 days in an incubator at 20°C. Oxygen is consumed and, as ammonia oxidation is inhibited, carbon dioxide is released. The total amount of gas, and thus the pressure, decreases because carbon dioxide is absorbed. From the drop of pressure, the sensor electronics computes and displays the consumed quantity of oxygen. The BOD value is most commonly expressed in milligrams of oxygen consumed per liter of sample during 5 days of incubation at 20°C and is often used as a robust surrogate of the degree of organic pollution of water.

The major nutrient concentrations of the wastewater (Nitrite-N, Nitrate-N, Ammonium –N, phosphate–P including total nitrogen and total phosphorus) concentration was determined spectrophotometrically. In all the cases the concentration of nutrients in the sample is determined by the addition of a reagent to the raw water sample. After allowing time for color development, the color is read at the wavelength between 400-500 nm.

Biomass concentrations were determined by weighing dried (24 h, 105°C) 100 ml samples of the liquid phase. 100 ml of a well mixed sludge is pipette to a suction filtration apparatus and the excess water is removed. Then the sample on the pre weighed filter paper transferred to preweighed dish and evaporated to dryness in a drying oven. The dish is cooled in desiccators to balance temperature, and weighed.

A spectrophotometer (Hach Lange Xion 500 LPG385) was employed for the measurement of COD and nutrients. A thermostatically controlled incubator with standard/glass door was employed during the measurement of BOD_5 to control temperature. Microprocessor-controlled standard-pH-ion-meter pMX 3000/pH was used to measure the pH values and temperatures of the influent and effluent. The influent and effluent samples collected and kept in a refrigerator were analyzed for the selected parameters using the Dr.Lange cuvette test system. During all experiments, COD concentration, pH, NH_4-N, PO_4-P, TN and temperature measurements were made three times per week unless otherwise indicated. COD removal efficiencies of the trickling filter was calculated based on the reduction of COD concentration between the influent and effluent streams as shown in Eq.(1).

$$\text{COD removal (\%)} = (C_{in} - C_{out})/C_{in} \times 100\%, \quad (1)$$

where C_{in} is the influent COD concentration (mg/L) and C_{out} is the effluent COD concentration (mg/L).

3.2.6 Control of excess sludge

As backwashing is not suitable for this type of bioreactor, to control the sludge the trickling filter was washed by pumping 0.1 mol/L NaOH solution repeatedly for the first time after the trickling filter operated for about 22 days, then it was washed every two weeks. To bring the microbial environment to near neutral, the reactor was subsequently washed with tap water. The other operation done on the trickling filter to reduce excess biomass was trickling filter starvation for a few days by turning off both the influent flow and recirculation flow for 3 to 4 days.

3.3 Results and discussion

3.3.1 Real wastewater characteristics and preparation of synthetic water

The detailed characterization that was carried out on the Rostock brewery wastewater is shown in Table 3.2. The analysis characterized the brewery wastewater as having BOD_5, COD, NH_4–N, NO_2–N, TN, PO_4^{3}–P and TP concentrations ranges of 1.412–3.980, 1.651–9.306, 0.003–0.011, 0.229–0.440, 0.010–0.101, 0.008–0.01, and 0.017–0.050 (g/L), respectively. All PO_4^{3}–P, NO_3–N, NO_2–N, and NH_4–N values stated throughout are soluble values only and were determined from micro filtered samples (0.45 µm).

Table 3.2 Physicochemical characteristics of the brewery wastewater

Items	Unit	Maximum	Mean	SD	Minimum
COD	mg/L	9 303	5 351	3 228	1 651
BOD_5	mg/L	3 980	2 101	1 015	1 342
NH_4-N	mg/L	10.50	4.96	2.94	2.47
NO_3-N	mg/L	25.00	15.00	14.30	4.90
NO_2-N	mg/L	0.44	0.25	0.14	0.09
TN	mg/L	102.0	47.4	40.10	7.20
PO_4-P	mg/L	10.12	6.34	4.20	0.36
TP	mg/L	43.40	25.42	13.69	13.00
Temp.	°C	34.00	25.00	6.40	17.00
PH	pH-unit	12.00	10.00	1.60	6.00

This chapter deals with the beginning of phase one investigation in which a synthetic sewage (Boeije, 1996) was used. The type and concentration of each ingredient is based on typical concentrations of BOD_5, COD, pH and nutrients found in brewery wastewater (Driessen and Vereijken, 2003) as well as on the analysis of samples obtained from the local brewery, Germany. Synthetic brewery wastewater was composed of 1 g/L malt extract, 0.5 g/L yeast extract, 0.15 g/L peptone, 0.86 g/L maltose, 0.10–2.20 g/L $(NH_4)_2SO_4$, and 2.80 ml/L ethanol. Buffering salts of 0.08 g/L NaH_2PO_4 and 0.14 g/L of Na_2HPO_4 are added to maintain the pH at 6.7. The resulting synthetic wastewater was characterized for COD, N and P colorimetrically.

The theoretical COD of the ingredients was calculated from their oxidation equation. For yeast extract, malt extract, maltose, and peptone, it is assumed that 1 mg/L of product equals 1 mg/L of COD. This was verified by comparing the calculated and the measured total COD of the influent (Boeije, 1996). According to one chemical supplier in India (Jeevan Chemicals and Pharmaceuticals) the measured value of N in peptone and yeast extract is 10.0% and 10.5%, respectively. And as per another supplier in United Kingdom (Murphy & Son Limited), typical malt has an analysis for TN within 1.45%–1.75%. The calculated COD: N: P ratio of the synthetic influent was 102:2:1 (w/w/w). The theoretical BOD_5, assuming a COD to BOD_5 conversion factor of 0.65, was 3451 mg/L. The composition of the synthetic influent based on this fact is presented in Table 3.3.

Table 3.3 Composition of the synthetic influent used in this study

Name	Formula	Concentration [mg/L]	Solution
Ammonium sulphate	$(NH_4)_2SO_4$	2200	Nitrogen source
Disodium hydrogen phosphate	Na_2HPO_4	140	Phosphorus source & buffer
Ethanol	CH_3CH_2OH	2.8 mL/L	C-source
Malt extract		1000	C-source
Maltose		860	C-source
Peptone		150	C-source
Sodium hydrogen Phosphate	NaH_2PO_4	80	Phosphorus source & buffer
Yeast extract		500	C-source

3.3.2 Trickling Filter Operation and Performance

The pilot scale trickling filter was first seeded with real brewery wastewater. Near neutral brewery wastewater sludge (pH 6.7) was obtained from the local brewery. The wastewater was collected from the bottom of the chamber and taken with a cleaned plastic Jeri Can having a volume of 10 L. The reason to seed the trickling filter with this effluent is due to the fact that there is a high probability of getting microbial populations that are familiar to the wastewater (Rittman and Whiteman, 1994; Leta, 2004).

After the trickling filter seeding with the real wastewater and before developing performance data, the acclimation of the trickling filter was carried out. At the beginning trickling filter was operated for seven consecutive days with a mixture of real and synthetic water by feeding new wastewater on the 1st, 3rd, 5th and 7th days. In all cases, the total COD concentration of the influent was 2 000 mg/L. Meanwhile monitoring of effluent COD concentration was carried out. COD removal efficiencies of 81.7%, 71.5%, 79.8% and 86.8% were achieved as monitored on the 1st, 3rd, 5th and 7th days, respectively. The slight decrease in efficiency between 1st and 3rd day (81.9 % to 71.55 %) is due to the inclusion of a portion of the synthetic brewery wastewater, because the biofilm on the trickling filter medium was originated from real brewery wastewater. Then the efficiency start to increase again (71.55% to 86.8%) as the biofilm start to adapt the synthetic water. Then the acclimation of the trickling filter continued only with synthetic wastewater from the 7th day onwards. This time the influent COD was varied from about 1 000 to 4 800 mg/L. Here also the efficiency slightly decreased (86.8% to 74%) at the beginning. Then after COD removal efficiency ranging from 74.0% to 95.4% was achieved. At this point, we considered the start up period completed and the filter ready for full operation. The trend of COD degradation during acclimation phase of the trickling filter is presented in Fig. 3.4.

3.3.3 Efficiency of the trickling filter treating synthetic brewery wastewater at different hydraulic loading rates (HLRs) and organic loading rates (OLRs)

HLRs and OLRs are the major operational control factor for improving treatment efficiency of trickling filters. Therefore the response of the trickling filter for different HLRs and OLRs was investigated. To investigate the effect of HLRs, four different flow rates were used in the feed, namely 800, 700, 600 and 500 L/d including recirculation flow at constant influent COD concentration. The system efficiency ranged 77.70% – 93.10%, 79.40% – 96.70%, 88.30% – 91.20% and 84.10% – 88.30% COD removal for the hydraulic loading of 6.3, 5.6, 4.8 and 4.0 $m^3/(m^2 \cdot d)$ respectively as a result. When considering the effect of hydraulic loading rate, the phenomenon that makes the efficiency of the trickling filter to vary are the duration of the wastewater in contact with the bio film. It took for the wastewater to trickle through the trickling filter and reach the under drain from 5–20 minutes for a single pass as recorded during the operation of the trickling filter. Under normal circumstances the efficiency of the trickling filter decrease with increase in hydraulic loading rate due to reduced contact time of the wastewater with the bio film.

However increasing hydraulic loading rate could also increase the efficiency of the trickling filter as it further increase new microbial growth. Increasing hydraulic loading rate has also control over biofilm thickness there by decreasing substrate diffusion limitation. Nevertheless in this investigation there was not any significant variation in efficiency. The trend of COD reduction and COD removal effeciencies with the change in hydraulic loading is shown in Fig.s 3.5 and 3.6. And raw data tables are given in Annex A1 and Annex A2.

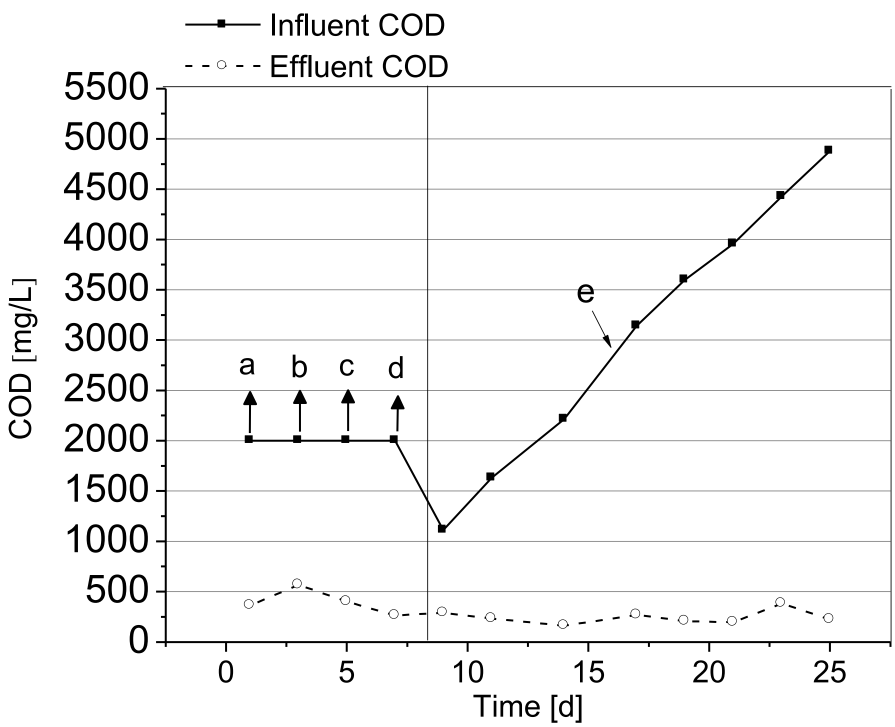


Figure 3.4 **Acclimation behavior of the trickling filter for COD degradation; a=75% real and 25% synthetic, b=75% real and 25% synthetic, c=50% synthetic and 50% real, d=25% real and 75% synthetic and e=100% synthetic**

Linear regression model that predicts the mass removal rate as a function of the mass loading rate is given by y = 8.76+0.8958x, where x and y stands for mass loading rate and mass removal rate ($g/m^2/d$) (Figure 3.7). The slope of 0.8958 means that when the mass loading rate increased by 1 $g/(m^2 \cdot d)$, the mass removal rate increased accordingly by a factor of 0.8958. More over the R-square value also confirms that there is a direct relationship between the two parameters with 99% confidence limit.

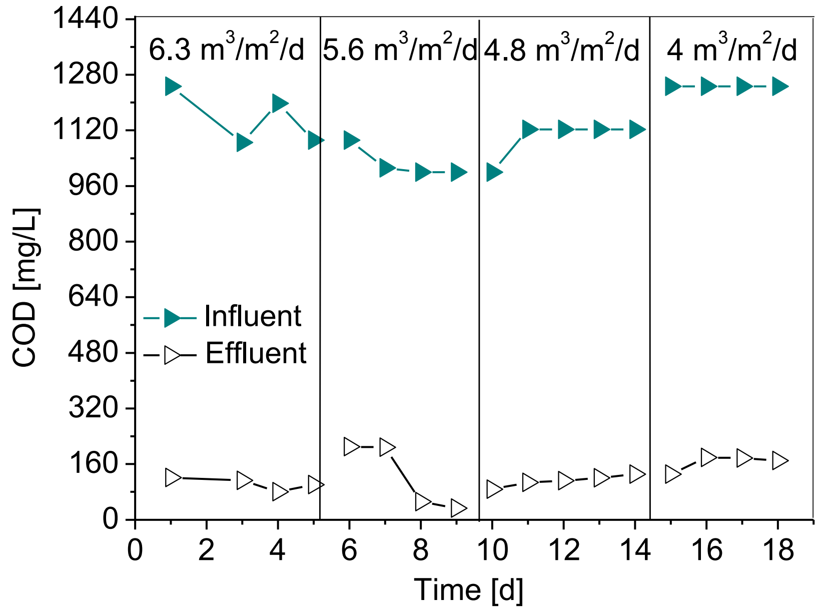


Figure 3.5 The trend in COD reduction as a function of hydraulic loading rate

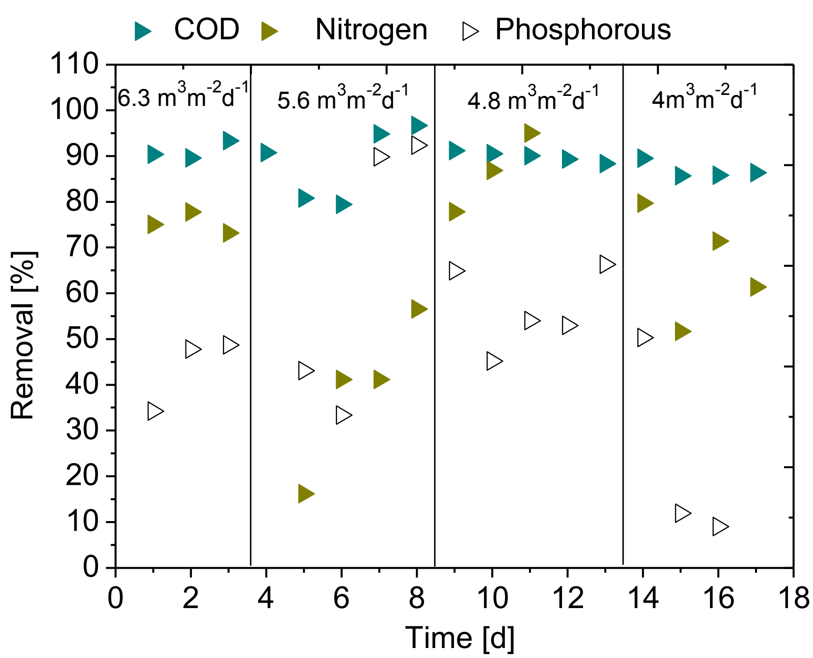


Figure 3.6 The trend in the performance of the trickling filter as a function of hydraulic loading rate

The response of the trickling filter for different organic loading rates was studied by varying the initial influent COD concentration (COD in the feed was increased to 1 000, 1 500, 2 000 and 3 000 mg/L) for a constant hydraulic loading of 6.3 $m^3/(m^2 \cdot d)$ and organic volumetric loadings of 1.5, 2.5, 3.0 and 4.5 kg $COD/(m^3 \cdot d)$ respectively. The operation of the trickling filter at each organic loading rate was repeated till nearly stable effluent COD is achieved. The trend of removal of COD is as illustrated in Fig. 3.8. And the raw data table is given in Annex A3. Where as Fig. 3.9 illustrate the performance of the trickling filter at the different organic loading rate. The percent COD removal efficiencies (mean ± SD) at each organic loading rate were 87.1 ± 3.3, 85.86 ± 5.7, and 85.1 ± 12.7 and 80.4 ± 3.5 respectively. The result of this operation also indicates there is only very slight decrease in efficiency of the trickling filter for the given range of organic loading rates. However the effluent COD level was high at organic loading rates of 3.0 kg $COD/(m^3.d)$ and 4.5 kg $COD/(m^3 \cdot d)$, beside this there is a problem of excess sludge accumulation rapidly at these organic loading rates. The recommended design COD loading for the present trickling filter is therefore 1.5 kg $COD/m^3/d$.

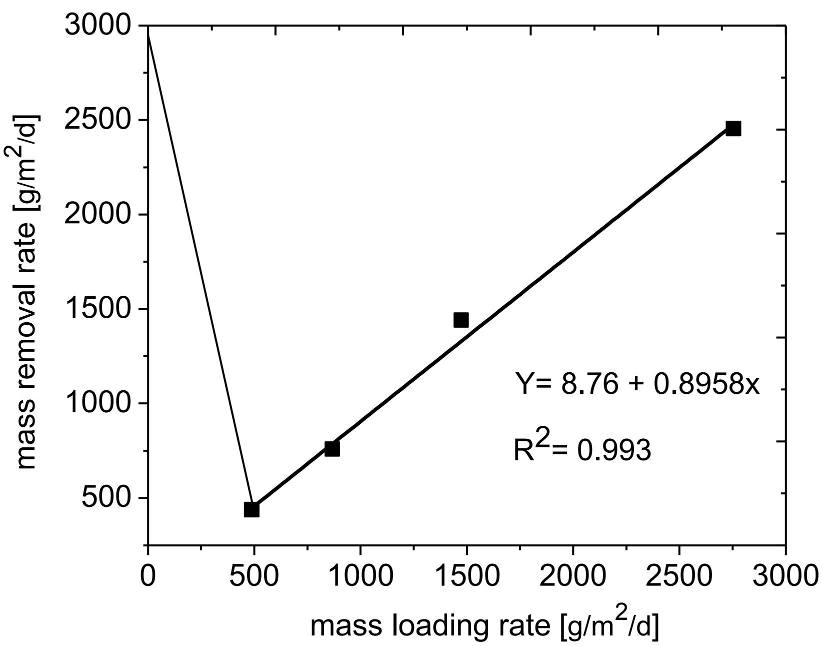


Figure 3.7 Relationship between mass loading rate and mass removal rate as a function of hydraulic loadings (not-including recirculation)

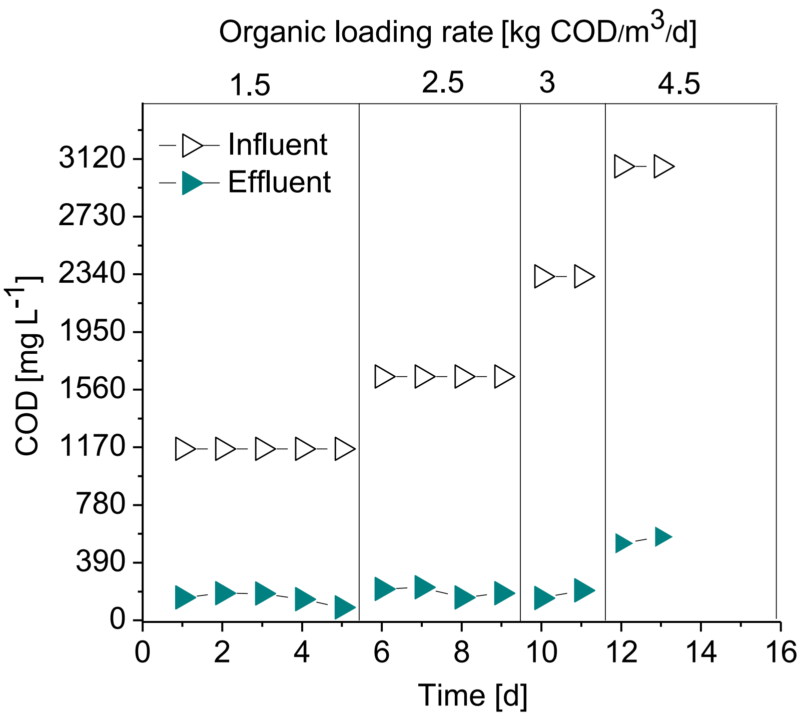


Figure 3.8 The trend in the removal of COD at different COD loadings

The quantity and quality of brewery waste water fluctuates significantly, depending upon operations like raw material handling, wort preparation, fermentation, filtration, controls in process (CIP) and packaging. Therefore in this investigation the performance of the trickling filter by varying the flow rates and influent COD concentration was also studied. Four different flow rates were chosen namely: 300, 250, 200, and 150 L/d. In the previous section of this chapter the investigation of effect of organic loading rate at the optimized inflow was investigated. In this section the investigation is the effect of the organic loading rate at different flow rates and viceversal. The trend of the trickling filter effluent concentration reduction during each run was then monitored. It is found that there is slight increase in efficiency with influent COD concentration in all flow rates. Increased microbial growth and activity with increase in COD loadings is the most probable reason. However an important conclusion that can be drawn here is that the performance of the trickling filter only slightly decreased with the changing flow rates and COD concentration of the brewery wastewater. As can be seen from Fig.3.10 the COD removal efficiency of the trickling filter was above 80% in all cases.

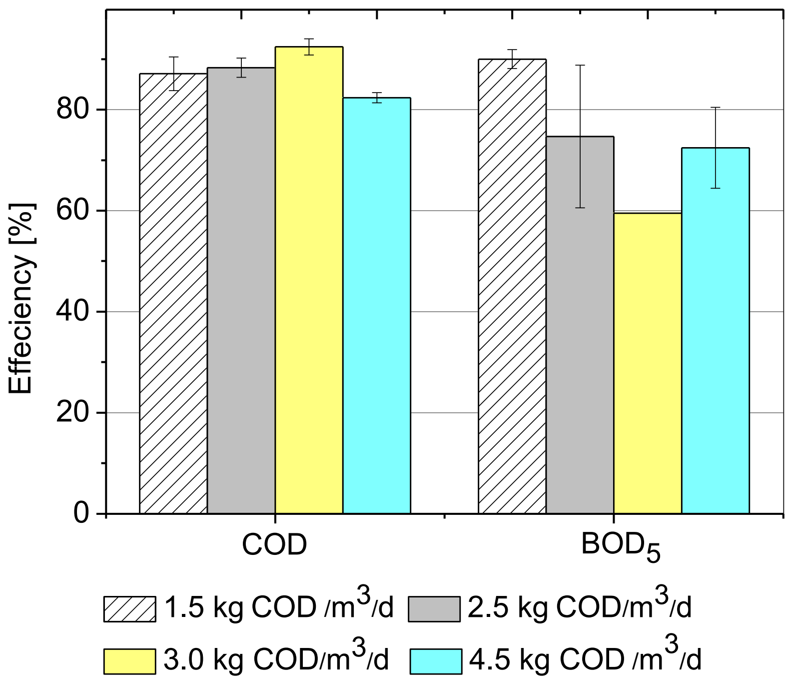


Figure 3.9 The performance of the trickling filter at the different organic loadings

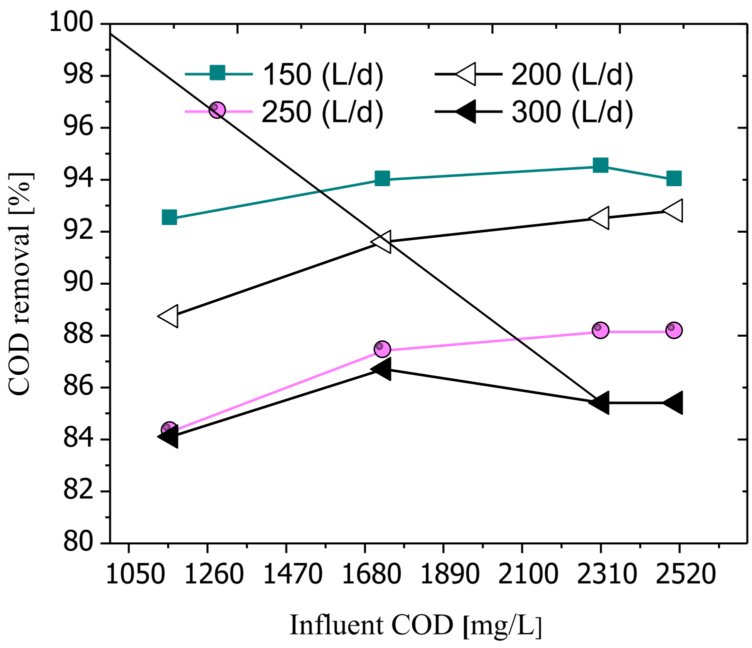


Figure 3.10 Effects of wastewater COD concentration and flow rate on the efficiency of the trickling filter

3.3.4 Effect of temperature and pH on efficiency of the trickling filter treating synthetic brewery wastewater

Temperature is very important parameter during the assessment of the overall efficiency of a biological treatment process. Temperature influences the metabolic activities of the microbial population and has also a profound effect on such factors as gas-transfer rates and the settling characteristics of the biological solids (Metcalf and Eddy, 1991a; Crites and Tchobanoglous, 1998). Temperatures below the optimum typically have a more significant effect on growth rate than temperatures above the optimum. It has been observed that growth rates double with approximately every 10°C increase in temperature until the optimum temperature is reached (Metcalf and Eddy, 1991b).

The performance of trickling filters will be affected by changes in the temperature of the filter films and the liquid passing over the films. It is usually assumed that these two temperatures will be essentially the same when only the aerobic portion of the film is considered. A decrease in temperature results in a decrease in respiration rate, a decrease in oxygen-transfer rate and an increase of aerobic film at a lower activity level, yielding a somewhat reduced efficiency at lower temperatures. Mathematically the relationship of efficiency and temperature can be expressed as (Onda et al., 1968)

$$E_T = E_{20} \times 1.035^{(T-20)} \tag{2}$$

where E = filter efficiency and T = temperature [°C].

This investigation was therefore aimed at ascertaining the effect of temperature in organic and nutrient uptake abilities of the bacterial species in the trickling filter. To investigate the effects of temperature on the trickling filter performance, Julabo FP40-MC refrigerated heating circulator is used to transfer heat between influent and heated water fluid streams. The working temperature of the temperature control device was set at 20°C, 25°C, 30°C, 35°C and 40°C. The wastewater kept at each temperature for 3 quensiquitive days unless other wise indicated. This temperature range was chosen to test the performance of the trickling filter, because it is the typical range for the wastewater of most brewing companies (Driessen and Vereijken, 2003).

The bacterial species that has grown on the trickling filter medium was able to grow and degrade the COD in the wastewater in the temperature range of 20°C to 35°C. Average degradation of COD (86.12% and 86.89%, respectively) was observed at an initial wastewater temperature of 20°C and 25°C. An increase in temperature of the wastewater increased the degradation rate of COD till temperature value of 35°C. The highest degradation of COD (89.2%) was noted at an initial wastewater temperature of 30°C.

The performance of the trickling filter for the removal of COD is greater than 80% at all temperature values and there was only small variation in the performance indicating, temperature only exhibits a minor effect on the performance of the trickling filter for the given temperature range. The trend in the removal of COD at each temperature is illustrated in Fig. 3.11 and Fig. 3.12 illustrates the effeciencies of removal at each temperatures.

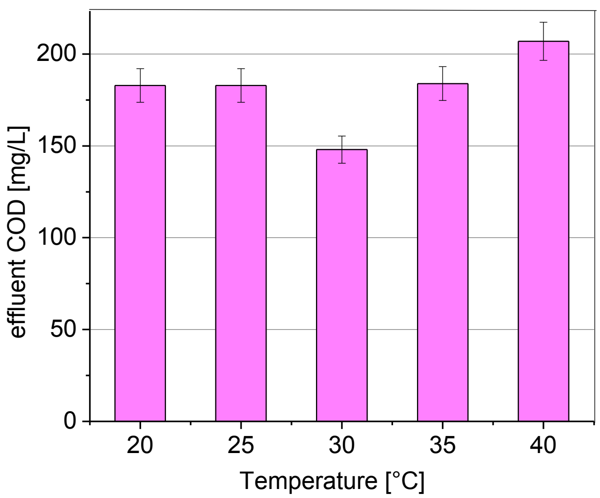


Figure 3.11 Trend in the removal of COD at the different temperature, COD_{feed} =1296.64 ± 78.88 mgL^{-1}

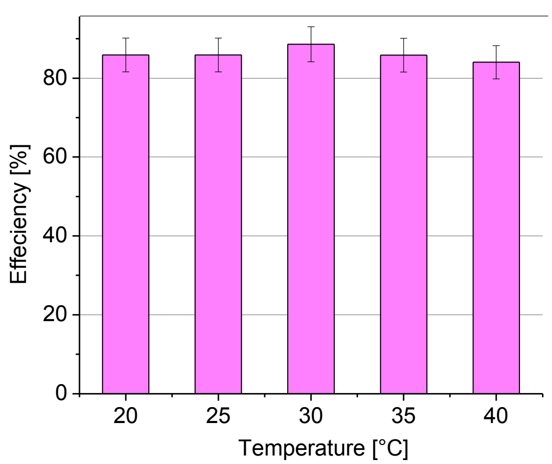


Figure 3.12 Effeciency of COD degradation at the different temperature, COD_{feed} =1296.64 ± 78.88 mgL^{-1}

The effect of pH on COD degradation capacity of the trickling filter was also examined over a pH range of 5.00–9.00. The pH was adjusted manually to the desired value by the addition of 0.1 mol/L NaOH and 0.1 mol/L HCl as necessary and checked every 30 min. The trickling liquid at each specific pH was kept for two consecutive days unless otherwise indicated. Fig. 3.13 depicts the trend of COD reduction by the trickling filter measured after each run. The removal efficiency was maintained at a high value, (86.67±9.9) % between pH 6 and 8 and around 94.6% at pH 6.3 where as removal capacity increased from 66.13 to 81.53 g COD/m^3/h when decreasing the pH from 8.00 to 6.3 and dropped to values of 46.63, 54.9 and 55.92 (gCOD/m^3/h) for pH values of 5.10, 8.3 and 9 respectively (Figure 3.14). The optimal degradation capability was obtained when the pH was regulated at a value of 6.3. This may be due to two reasons: on the one hand, it has been shown that different autotrophic and heterotrophic microbial groups and activities dominate at different pH values; on the other hand, the degree of availability of the different substrates is different at different pH values in the wet bio film where the biodegradation takes place. To conclude, unlike temperature effect there is a significant decrease in efficiency at low and high pH values. The trickling filter performs best in the pH range of 6.3–7.0, indicating that there might be a need to adjust the pH of the wastewater to near neutral to maintain the high effeciency.

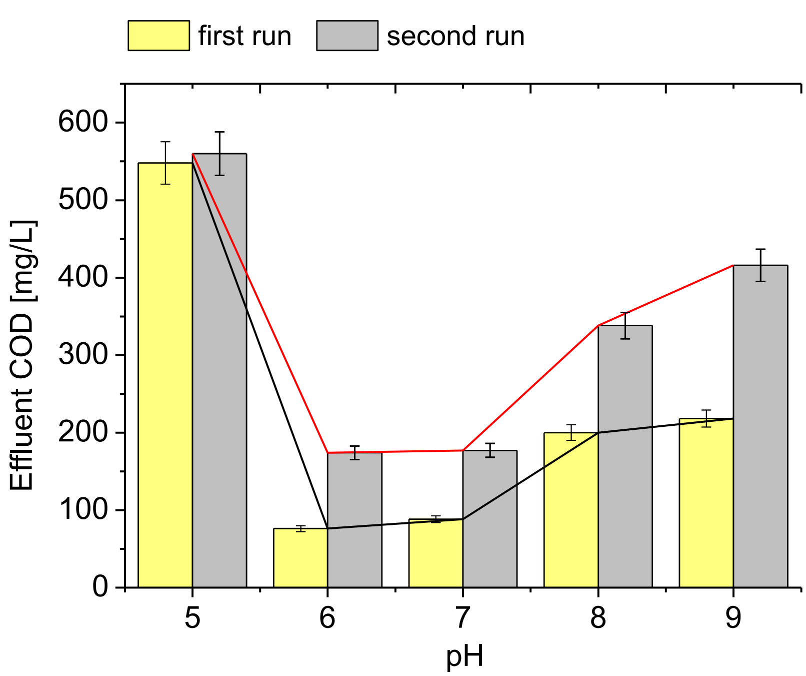


Figure 3.13 Trend in the removal of COD at the different pH, COD feed =1296.64 ± 78.88 mg/L

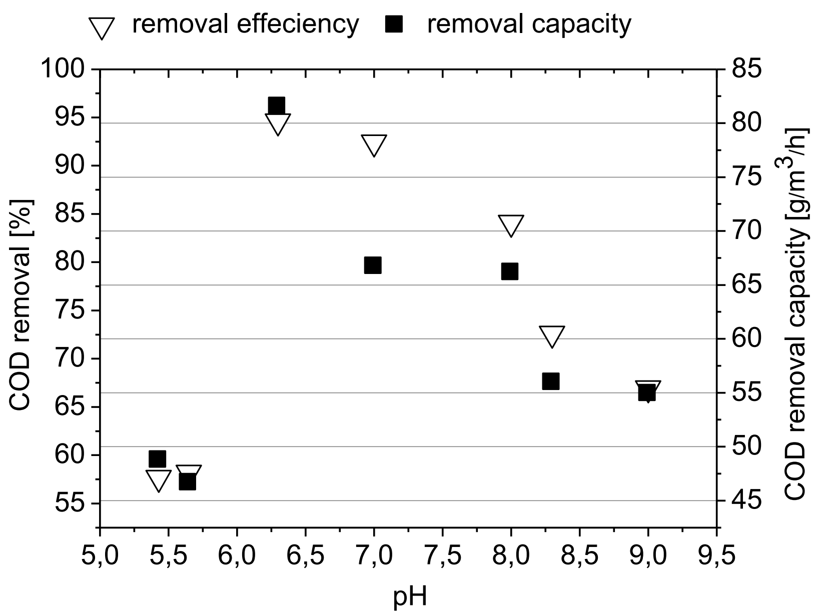


Figure 3.14 COD removal capacity and removal efficiency of the trickling filter as a function of initial wastewater pH; Inflow: 300 L/d; COD_{feed}: 1 102.59 ± 68.93 mg/L

3.3.5 COD removal profiles of the trickling filter

To take wastewater samples along the trickling filter height, five profile sampling ports at 52, 78, 104, 130 and 156 cm from the top were drilled through the pilot scale trickling filter wall and some cm into the media on the same side of the trickling filter. A tube with a 400 mm slot 300 mm from its end was inserted into these ports.

Profile of effluent COD concentration along the height of the trickling filter shows a rapid decrease in COD as the water flows from the top to the middle section of the trickling filter that indicates anoxic and anaerobic zones develop readily near the middle and the bottom section of the trickling filter medium. The occurence of high COD degradation in the top and middle section of the trickling filter can also lead to a conclusion that in the upper part of the trickling filter system; mainly COD was oxidized while nitrification could take place in the lower part of the system, where nitrifiers are available. The COD profile result also proves that there exist uniformity in bio film cover and substance transport along the depth of trickling filter. The average effluent COD at the top, middle and bottom section of the trickling filter for the different days of operation was 100.2 mg/L, 89.33 mg/L and 88.74mg/L respectively. Where as COD in the final effluent was in the range of 86-106 mg/L at influent COD concentration of

1286.88±50.35 mg/L. Figs. 3.15 and 3.16 illustrate the trend of COD profile with time and the average COD profile of the trickling filter respectively.

In Fig. 3.17 the prediction of COD removal efficiencies of the trickling filter by steady state trickling filter models as a function of depth is depicted. Out of the three models which are calibrated using the observed COD profile data, the modified form of Velz equation assumes a plug flow hydraulic regime with first order degradation of COD as it passes through the trickling filter, making COD removal as a function of hydraulic retention time. And experimentally found result also confirms this. The models predict a more gradual increase in efficiency with depth of the trickling filter than what is obtained experimentally. As a result at this loading of the trickling filter about a height of 0.6 meter is the optimum trickling filter height concerning COD removal. Whereas according to the models predictions the optimum height is 1.6 meter.

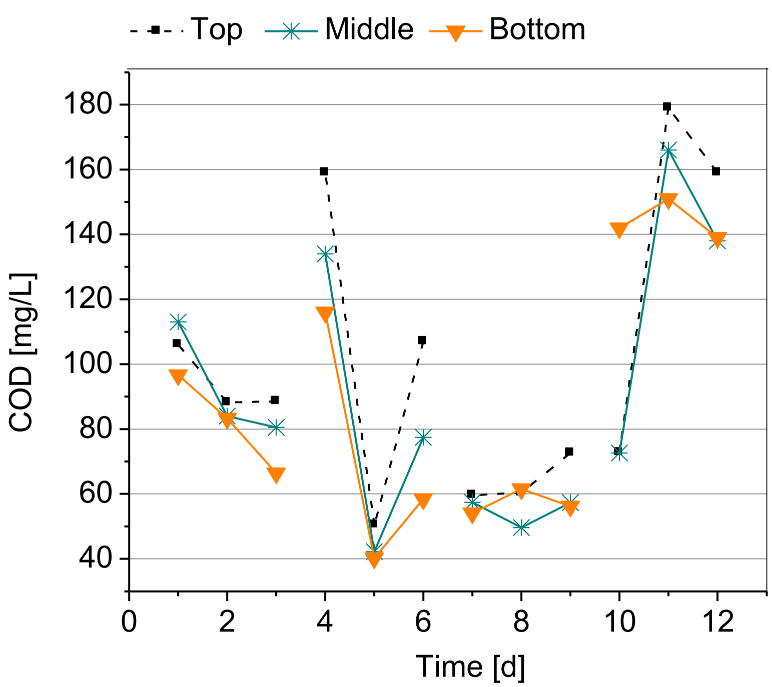


Figure 3.15 Trend of effluent COD profile of the trickling filter as a function of time, Inflow:300 L/d; Feed COD: 1286.88 ± 50.35 mg/L

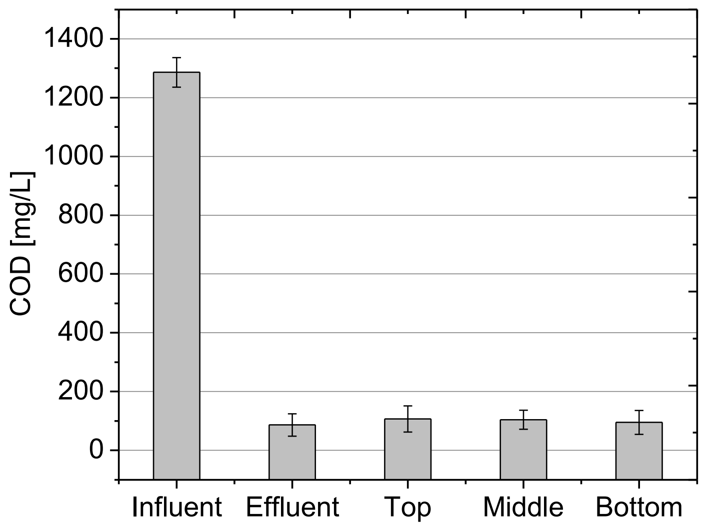


Figure 3.16 Average effluent COD along the trickling filter height.Inflow: 300 L/d; CODfeed: 1286.88 ± 50.35 mg/L

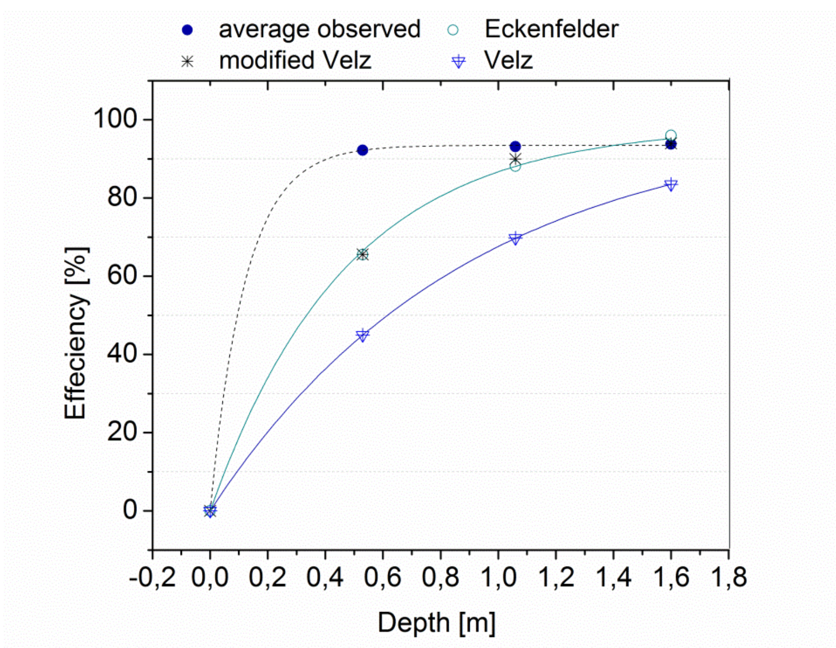


Figure 3.17 Average observed and predicted COD profile as a function of depth.

3.3.6 Sludge reductio

Sodium hydroxide washing

The operation of the trickling filter with COD removal efficiency (mean±SD) of (92±2.7) % was maintained until day 17. However, after day 18 due to excessive biomass accumulation, the removal efficiency was decreased. The removal efficiency was below 63 % on days 18, 19, 20 and 21, and 51.13 % at day 21. The major cause for the decrease in trickling filter efficiency is believed to be the reduction of the bio film-specific surface area with increases of biomass contents (Alonso *et al.*, 1997). To remove the excess sludge therefore, the trickling filter was washed by repeatedly pumping about10 L of 0.1 mol/L NaOH solution (Weber and Hartmans, 1995). There was no significant decrease in the efficiency of trickling filter due to change in pH as a result of NaOH washing. The higher COD removal efficiency of the trickling filter was regained after washing of the trickling filter, because of the higher specific surface area of the trickling filter for the attachment of new bio film and increase in porosity after the excess sludge is reduced. Fig.3.18 depicts the marked decrease in efficiency due to the excess sludge and maintenance of the higher removal efficiency after NaOH wash.

Few days starvation

Decreased efficiency as a result of excess biomass accumulation was also managed by starving the trickling filter for few days. The trickling filter was starved for a total of 4 d as described in Table 3.4. After starvation, there was a significant decrease in the biomass which may be the result of microorganism death, endogenous respiration, or secondary processes such as the predation of higher organisms (Zhang *et al.*, 2009). The recovery processes commenced after a 4 d starvation period, at which time normal operation of the bio trickling filter resumed. The bio trickling filter obtained high removal efficiency within 2-d of normal operation. Fig. 3.19 shows the efficiency of the trickling filter before and after starvation. The fast recovery of the bio trickling filter for COD removal suggests that there is no need for the build up of significant amounts of new degrading biomass to resume the normal operation of the trickling filter.

Table 3.4 Operating conditions for few days trickling filter starvation

Operation day	Influent flow	Recirculation flow supply	Atmospheric air supply	Possible conditions
1st & 2nd	on	off	on	Defect of recirculation pump, equipment malfunction,discontinuity of electricity
3rd & 4th	off	off	on	Time of no beer manufacturing (week end or holiday)

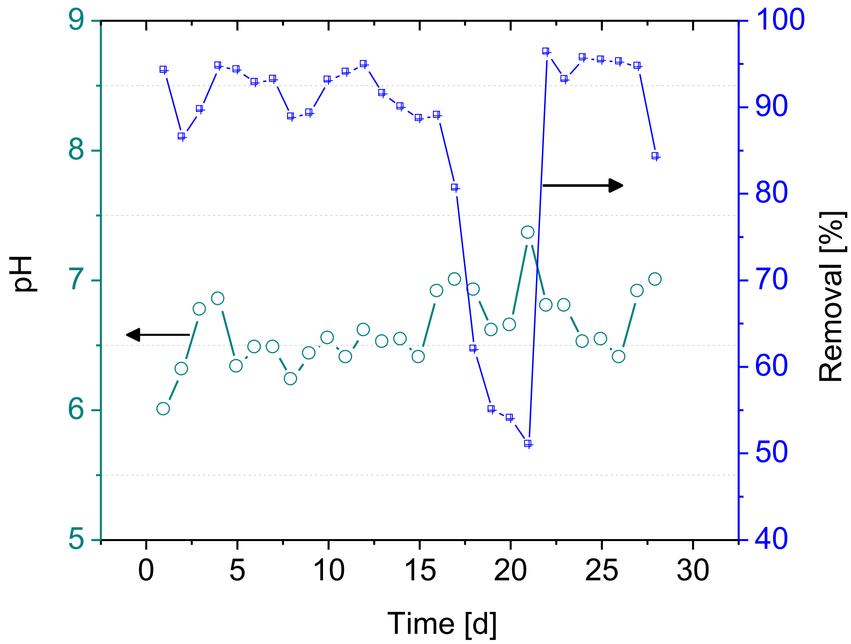


Figure 3.18 Drop in trickling filter COD removal efficiency due to bio film thickness and its recovery after NaOH wash

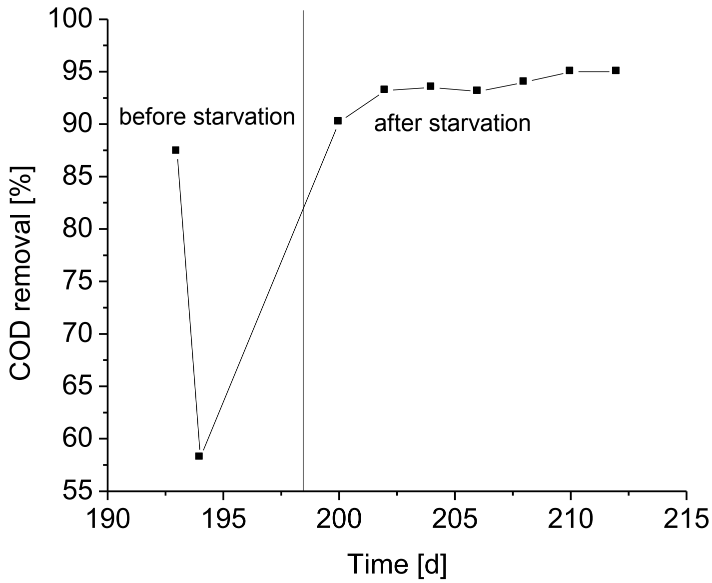
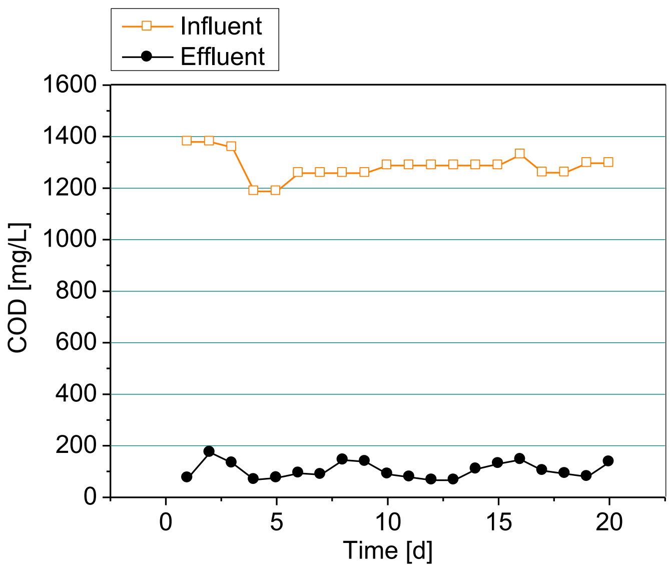


Figure 3.19 Drop in trickling filter COD removal efficiency due to bio film thickness and its recovery after starvation

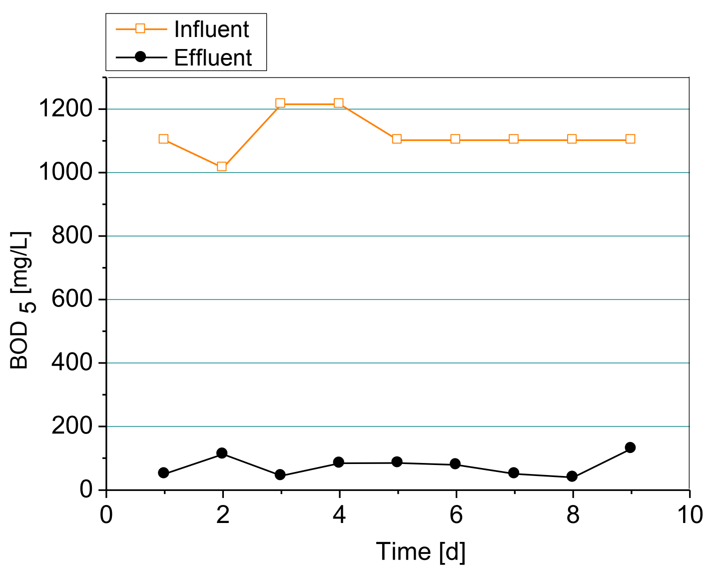
3.3.7 Steady state performance of the trickling filter

After the trickling filter was operated for a total of 6 months which includes its start up and its operation under variable loading conditions, then on the basis of efficiency and rate of sludge accumulation the optimum load, temperature and pH is selected. Then after the consistence of the performance was checked at the selected design loadings till a more or less constant performance is achieved and this was achieved after about three months of operation. Steady state of the trickling filter could occur when there is no change on the population of active COD degrading bacteria on the trickling filter. The major determining factors are quality and quantity of dissolved substances, the working pH, temperature and when new bacterial species grow and/or the existing bacterial species evolve into a new bacterial species with time. In addition to this the biofilm thickness dynamics also determine the achievement of steady state.

During the start up period the trickling filter possesses large filter media space which is available for the attachment of new bacterial biofilm. Therefore the performance of the trickling filter increase with time since the bio film cover increases as the trickling filter is operated longer and longer. In addition the bio film thickness is not large enough for detachment to take place. Then during the full operation of the trickling filter complete biofilm cover could occure in the trickling filter and the biofilm thickness might be large enough for the occurence of detachment. As a result the trickling filter will become far from steady state due to the dynamics of biofilm tickness. The steady state of the trickling filter is achieved when the trickling filter is operated at the optimized condition constantly and when the rate of bio film attachment is equal to the rate of detachment of the biofilm. In the present investigation consistence of removal effeciency is achieved with respect to COD and BOD_5 removal however the trend for nutrient removal tends to show some variation. The reason for the variation of phosphorous and nitrogen is discussed in detail in the next chapter. The trend of concentration with time (Figure 3.20) and the statistical summary of efficiencies at the steady state (Figure 3.21) are illustrated here under.



[A]



[B]

Figure 3.20 The trend in the removal of COD and BOD_5 at steady state, [A]& [B]

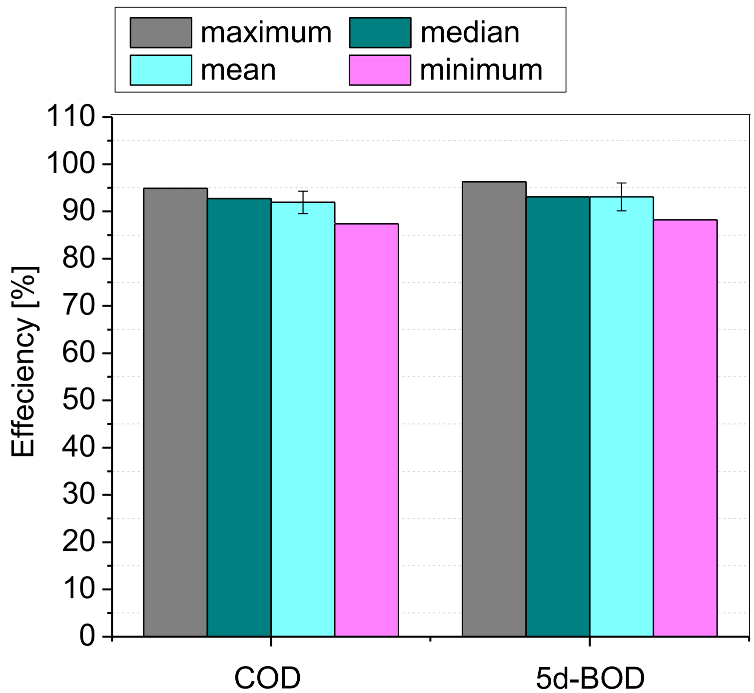


Figure 3.21 Performance of the trickling filter at its steady state

Conclusions

The development of microbial biofilm during start up on the trickling filter medium can be achieved with out a need for special inoculation. And a rapid start up period, only about 25 days was required before the trickling filter is ready for full operation. Increasing the hydraulic or organic loading rate has no significant effect on the efficiency. An average value of (84.42 ± 6.5) % COD removal efficiency could be achieved when the loading rate increased from 1.5 to 4.5 kg COD/(m^3·d). The trickling filter can have sufficient air circulation naturally and the only energy demand is for wastewater pumping. Handling of the excess sludge is not a problem because the amount of sludge produced is relatively small and as it is highly concentrated, it can settle easily inside the secondary clarifier. At its steady state the trickling filter could achieve a maximum removal efficiency of COD and BOD_5 removal (mean ± SD), 91.94 ± 2.38% and 93.10 ± 2.93% respectively at the design organic loadings of 1.5 kg COD/m^3/d and flow rate of 300 L/d. Therefore, the proposed biological treatment process appears to be a promising wastewater treatment method for the removal of COD from brewery wastewater.

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4 Verification studies of steady state trickling filter models for the removal of COD from brewery wastewater using the trickling filter

4.1 Introduction

Fundamentals of trickling filter process can be described by developing relationships among variables that affect trickling filter operation. And this is of course already done by numerious researchers. Many trickling filter process models are existed that ranged from simplistic empirical formulations to numerious models. Different trickling filter process operating data are analyzed to establish equations or curves to fit available data. And out of these data analysis several impirical trickling filter formulas have been developed. Unfortunately, numerious models exist but there is lack of an industry standard. Designers need to assess which equation best fits a particular situation when selecting a design model, especially with regard to the confidence level necessary to meet discharge permit requirements. Therefore many process designers use a forcasting approach and will apply several impirical models to evaluate a system. The following empirical models have been reported by Boltz et.al.(2009) and Boltz (2010) as options historically used to describe trickling filter performance in the context of process design, (1) national research council, (2) Velz (1948) equation, (3) Schulze (1960) equation, (4) Eckenfelder (1961) formula, (5) Galler and Gotass (1964), Germain (1966) equation, (7) Kincannon and Stovver (1982) and (8) the institution of water and environment management (1988) formula.

The aim of this chapter was therefore development of emperical formulas for COD removal performance of the present trickling filter using the existing trickling filter models.

Examination of the trickling filter models

Basically two approaches can be used for designing trickling filters namely, based on the performance of similar units which is summarized by a correlation to which National research council equation is an example:

$$E = \frac{1}{1+0.014\sqrt{\left(\frac{w}{V_m R_f}\right)}} \tag{1}$$

where E is efficiency of the trickling filter, w is BOD loading rate (kg BOD/d), V_m is total volume of filter media (10^3 m^3) and R_f is a recycle factor. Kinetic equations governing organic removal is the second approach to design a trickling filter. The first recognized attempt in this direction was made by Velz

(Bruce *et al.,* 1987) who suggested BOD (COD) removal is following a first order kinetics to BOD (COD):

$$\frac{dc}{dt} = -k_d C \tag{2}$$

where k_d is rate constant (m^{-1}). Upon various modification of the original Velz, the modified form of Velz equation becomes:

$$\frac{S'}{S_0} = e^{-\left[\frac{K_{20} A_S D \theta^{T-20}}{Q^n}\right]} \tag{3}$$

where S' is effluent COD concentration (mg/L), S_o influent COD concentration (mg/L), K_{20} treatability constant, A_s is specific surface area of the medium, D is depth of the trickling filter (m), θ temperature correction factor (1.035), Q hydraulic loading (m^3/m^2 (c.s area)d), and n is hydraulic flow exponent (0.5). This equation assumes a plug flow hydraulic regime with first order degradation of COD as it passes through the trickling filter, making COD removal as a function of hydraulic retention time. The kinetic constant in the modified Velz equation is either designated as a treatability coefficient, or simply as a K_{20} factor. Modified velz equation has a major constraint; the K_{20} value is different for different media types with the same specific surface area.

Eckenfelder (1980) suggested the inclusion of influent substrate concentration into the modified Velz equation in the denominator of the exponent of Eq.(3):

$$\frac{S'}{S_0} = e^{-\left[\frac{K_{20} A_S D \theta^{T-20}}{S_O Q^n}\right]} \tag{4}$$

It is also demonstrated by Eckenfelder (1980) that the fraction of COD removal could be described by the following equation based on volumetric organic loading:

$$\frac{S'}{S_0} = e^{-\left[\frac{K_{20} A_S D \theta^{T-20}}{L'}\right]} \tag{5}$$

Where L' is volumetric soluble organic loading (kg/m^3d). Temperature corrections factor to diffusivities were incorporated using a common correction term (James Welty *et al.,* 2008):

$$\theta_T = \theta_{20} \frac{T}{293} \mu_{20} \mu_T \tag{6}$$

Where the subscript 20 indicates the parameter value at 20°C (293K) μ is the fluid viscosity, and T is the temperature in Kelvin.

In another model, Stover–Kincannon model, the substrate utilization rate is expressed as a function of the organic loading rate using monomolecular kinetics for bio film reactors (Buyukkamaci & Filibeli, 2002). Kincannon and Stover suggested the use of the Monod like equation based on surface organic loading. This equation was known to be total organic loading model:

$$\frac{(S_0 - S')}{S_0} = \frac{U_{max}}{L'' + K_b} \tag{7}$$

where U_{max} biokinetic parameter (g/m^2(media area)d), L'' surface soluble organic loading (g/m^2(media area)d) and K_b biokinetic parameter (g/m^2(media area)d). In this model U_{max} represents the maximum removal rate of substrate and K_b is the constant of saturation value.

4.2 Materials and methods

The detailed description of the pilot scale trickling filter is given in chapter three. In this study, synthetic brewery wastewater with the characteristics given in Table 3.3 of chapter 3 was stored in the feed tank (1) and pumped to the top of the tower with the help of a peristaltic pump (2) a perforated plate arrangement (3) was provided at the top of the filter to distribute the wastewater uniformly. An under drain system was provided at the bottom of the tower to collect the treated wastewater. The synthetic brewery wastewater was prepared by adding the ingredients to tap water. Phosphate buffer and sufficient nutrients were added to support bacterial growth. COD in influent and effluent samples were analyzed by Hach Lange Laboratory analysis spectrophotometer. Hach Lange method determines the amount of oxidisable organic material in the sample by using potassium dichromate in sulphuric acid as oxidation agent. The sample is added into ampoules especially designed for this analysis. The color of the sample is then measured in a photometer after it has been heated at 176°C for 15 minutes.

4.3 Results and discussions

Fig.4.1 depicts the effect of influent COD concentration on COD removal at various hydraulic loading rates. Decreasing trend of COD removal with influent COD concentration and hydraulic loading rate was observed except for the hydraulic loading of 5.6 m^3/m^2 d. Reduced contact time of the wastewater with the bio film at higher hydraulic loading rate is the probable reason that has ultimate effect on COD removal percent. Logan et al (Bruce *et al.,* 1987) observed that the percentage of BOD removal decreased with hydraulic loading rate in the case of sewage treatment. In their study, the hydraulic loading rates in the range of 0.1±1.2 l/m^2 s were studied. The organic loadings in their study were in the range of 0.4 ± 0.9 kg/m^3. Fig.4.1 also illustrates the effect of initial COD concentration on the performance of the trickling filter. There was a gradual general decrease in COD removal with initial COD concentration. A study on treatment of municipal wastewater (95% domestic +5% industrial wastewater) made by Richards and Reinhart (Tyler Richards & Debra Reinhart,1986) using cross flow medium trickling filter of height 6 m also demonestrated that the BOD removal decreased with increasing BOD loading. Randall et *al*., (Andrew Amis Randall *et al.,* 1997) also observed almost a similar type of behavior in the case of synthetic fiber manufacturing wastewater using vertical flow medium trickling filter. However it is found from the linear regression model analysis that the mass removal rate increase with mass loading rate when both the hydraulic loading rate and organic loading rate increase (Figure 4.2).

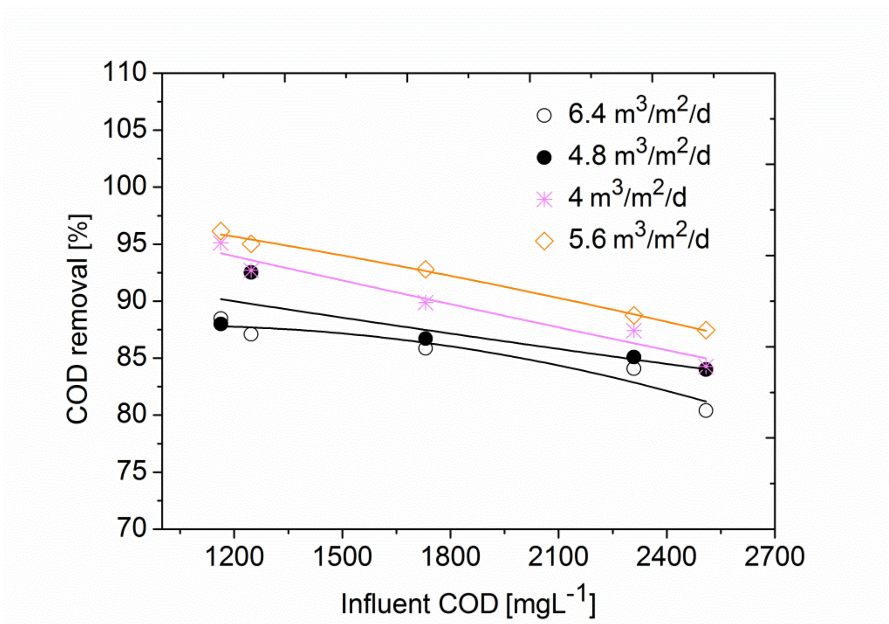


Figure 4.1 **COD removal performance of the trickling filter as a function of influent COD Concentration**

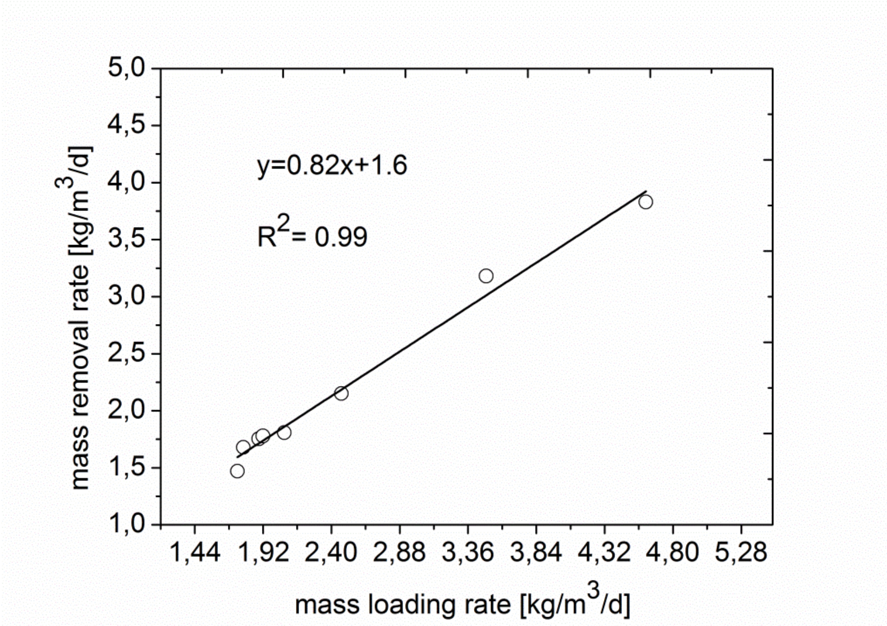


Figure 4.2 linear relationships between mass loading rate and mass removal rate

Curve-Fitting

The COD data generated using the aforesaid pilot scale trickling filter was evaluated using the four trickling filter models Eqs. (3, 4, 5 and **7**). In the first two models the hydraulic flow exponent is the function of the hydraulic characteristics of the media. And to provide a basis for comparison, a typical value of 0.5 (Tyler Richards & Debra Reinhart, 1986; Andrew Amis Randall *et al.,* 1997; Terry & Gayle, 1987) has been adopted here. The kinetic model parameters and the predicted effluent values are determined by minimizing the relative absolute error between model and measurement=sum (abs (model value-measured value)/measured value). Since the data is generated at temperatures varying around 20°C, the temperature correction factor in the first three models was ignored in calculating predicted COD removal. These predicted values in each model are then compared to the observed values to investigate the ability of the models to model the trickling filter performance. The results are discussed here under. The kinetic parameters estimated by modified Velz equation with the inclusion of influent COD concentration is given in Fig. 4.3 and for the other models data not shown for the sake of brevity. The kinetic parameters and R^2 values for the models are summarized in Table 4.1.

Fig. 4.4 [A] depicts the comparison of observed and predicted values of COD removal using modified Velz equation. The COD removal efficiency as a function of hydraulic loading rate as obtained from experimental values and model calculations are very close except the slight under estimation at hydraulic loading of 5 $m^3/m^2/d$. Likewise Fig. 4.4 [B] depicts the observed and predicted values of COD removal using modified Velz equation with the inclusion of influent COD. Due to the inclusion of the influent COD concentration, a slightly better agreement of the model is obtained as it can be concluded from their R^2 value. The comparison of the experimental COD removal with that of the Eckenfelder equation incorporating volumetric organic loading rate is depicted in Fig. 4.4 [C]. For the given range of organic loading rate the model and the experimental COD removal value are almost the same at the lower organic loading rates however, over estimation by the model was occurred at the higher organic loading rates. Similarly model estimation of the experimental COD removal with that of kinkannon and Stover model is shown in Fig.4.4 [D]. In this model the COD removal prediction was nearly the same as what is found experimentally in the entire organic loadings. In general all the examined models have good agreement with the present experimental COD removal data. Out of the four models modified velz equation with the inclusion of influent COD is found to be the best fit model. To evaluate the model performance, descriptive statistics of the residual errors are given in Table 4.2.

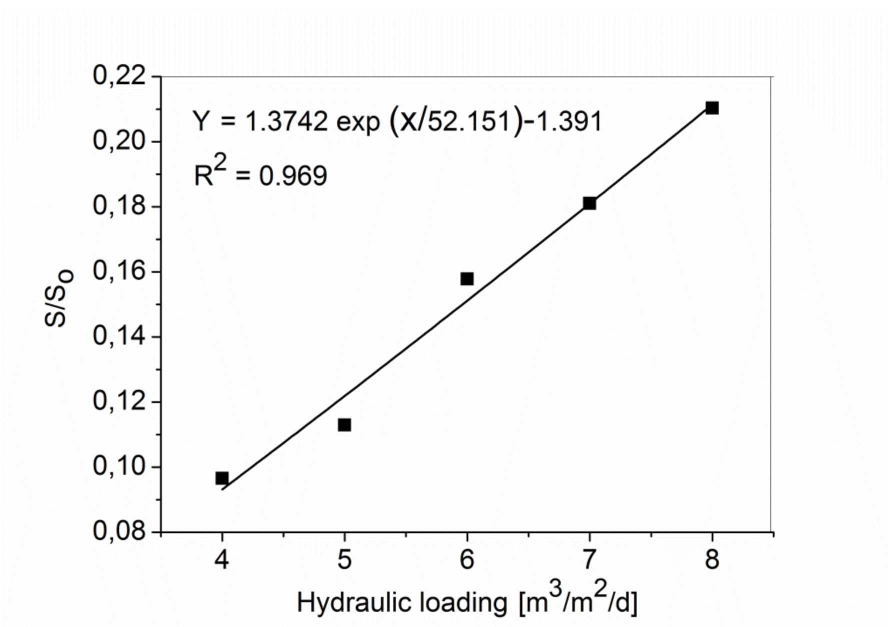
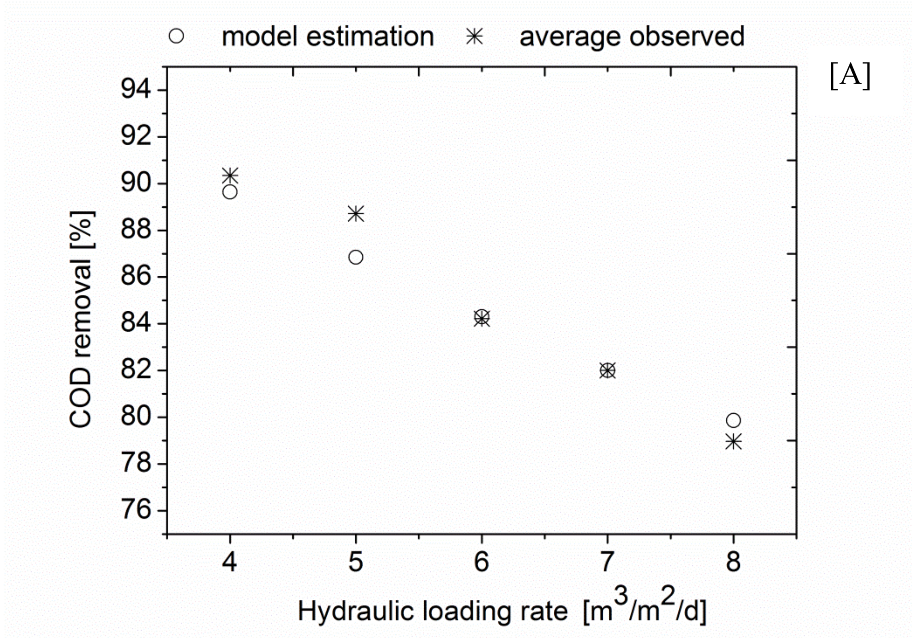
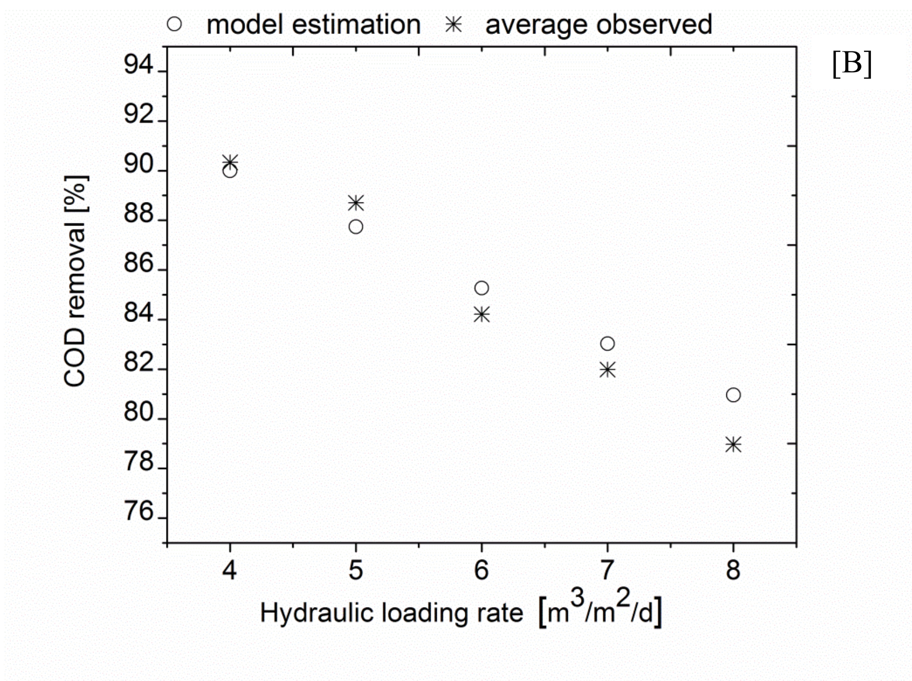


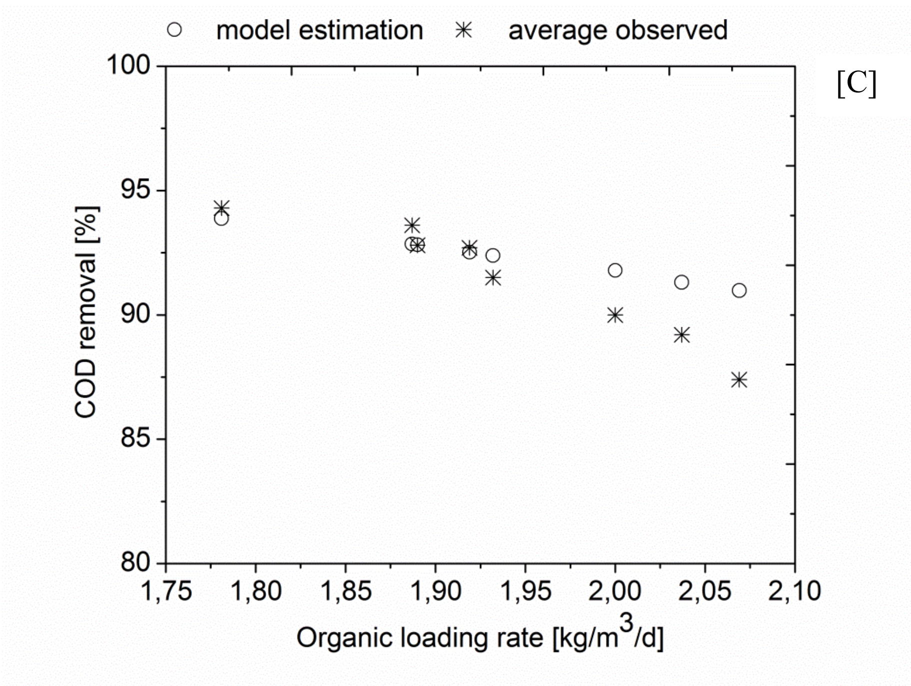
Figure 4.3 Estimation of kinetic parameters for modified Velz equation



○ model estimation ✳ average observed
[A]
COD removal [%]
94
92
90
88
86
84
82
80
78
76
4
5
6
7
8
Hydraulic loading rate [m3/m2/d]



○ model estimation ✳ average observed
[B]
COD removal [%]
94
92
90
88
86
84
82
80
78
76
4
5
6
7
8
Hydraulic loading rate [m3/m2/d]



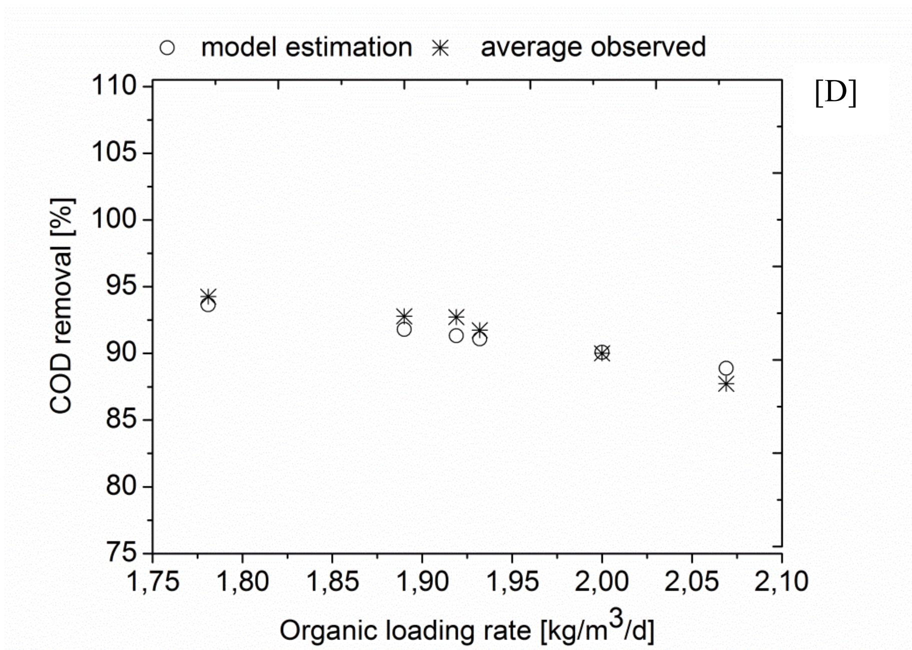


Figure 4.4 Curve fitting effectiveness of the trickling filter models using modified Velz, the modified Velz with the inclusion of influent COD concentration, Eckenfelder& Stover and Kincanon,[A] to [D] respectively

Table 4.1 Values of kinetic parameters

Design equation	Kinetic parameters	R^2
Modified Velz	k_{20} = 0.039 $[m^3/m^2.d]^{0.5}$	0.92
Eckenfelder's modified Velz	k_{20} = 42 $g/m^{2.5}.d^{0.5}$	0.96
Eckenfelder	k_{20} = 0.035 $kg/m^2.d$	0.78
Kincannon and Stover	U_{max} = 14 $g/m^2.d$	0.90
	K_b =10 $g/m^2.d$	

Table 4.2 Descriptive statistics of the residual errors

Residual statistics	Calculation	Regression results			
		modified Velz equation	Eckenfelder's modified Velz equation	Eckenfelder's equation	Stover–Kincannon equation
Residual tolerance	$(Ya - Yp)$	8×10^{-4}	8.2×10^{-4}	1.48×10^{-4}	8.11×10^{-4}
Sum of residuals	$\sum_{i=1}^{n}(Ya - Yp)$	3.4×10^{-2}	1.52×10^{-1}	2.5×10^{-2}	6.78×10^{-2}
Average residual	$\frac{\sum_{i=1}^{n}(Ya - Yp)}{n}$	7.5×10^{-3}	3.05×10^{-2}	3.1×10^{-3}	8.48×10^{-3}
Residual or error sum of squares(absolute)	$SSE = \sum_{i=1}^{n}(Ya - Yp)^2$	4.1×10^{-4}	1.3×10^{-2}	2.2×10^{-3}	8.24×10^{-4}
Residual or error sum of squares(relative)	$SSE_R = \sum_{i=1}^{n}\left[(Ya - Yp)^2 \frac{1}{\delta^2}\right]$	4.1×10^{-4}	1.3×10^{-2}	2.2×10^{-3}	8.24×10^{-4}
Standard error of estimate	$\sqrt{\frac{\sum_{i=1}^{n}(Ya - Yp)^2}{n - p}} = \sqrt{\frac{SSE}{n - p}}$	2.1×10^{-2}	8.1×10^{-2}	2.1×10^{-2}	1.28×10^{-2}

4.4 Summary and Conclusion

Brewery wastewater contains high COD concentrations which need to be removed before being discharged into the environment. It is reported in the previous chapter that trickling filter, which is the cheapest and environmentally friendly wastewater treatment method in comparisons to the other biological methods can effectively remove COD from brewery wastewater. And in this chapter the fundamental trickling filter design models are used to evaluate the ability of the models to model the performance of the trickling filter. Except Eckenfelder model all the other models predict the efficiencies of the trickling filter impressively very close to the experimental COD removal data. When all the four models are compared, modified Velz equation with the inclusion of influent COD concentration as suggested by Eckenfelder is found to be the best fit model for the present investigation. The COD removal at any influent COD concentration, S_o and hydraulic loading of Q can be calculated as S'/S_o = 1.3742exp (Q/52.151) -1.391, where S' is the effluent COD concentration.

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5 Performance of a trickling filter for nitrogen and phosphorus removal with synthetic brewery wastewater in trickling filter bio film

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Abstract

The aim of this investigation was to evaluate the nutrient removal efficiency of the trickling filter during the treatment of brewery wastewater. In this investigation the performance is evaluated under variable loadings and operating conditions and at constant loadings. The bioreactor's average efficiency ranged from 65.46 % to 86.59 % and from 10.45 % to 56.66 % for total nitrogen (TN) and total phosphorous (TP) respectively as the flow rates changed from 900 to 1100 L/d and at influent COD concentration of 1000 mg/L in which the average influent nitrogen and phosphorus concentration was 36.9 mg/L and 30.74 mg/L respectively. Mass removal rate increased with mass loading rate as can be proved from the linear regression model, correlation coefficients 98.2 % and 95.3% for nitrogen and phosphorous respectively. Change in COD content of the brewery wastewater had only little effect on the nutrient removal performance of the trickling filter. The trickling filter achieved nitrogen and phosphorus removal efficiencies (mean ± SD) of 70.6 ± 21.5 and 68.87 ± 8.52 respectively at COD load and flow rate of 1163 mg/L and 300 L/d. When the trickling filter achieved steady state with respect to COD removal total nitrogen and total phosphorus removal efficiencies were 88% and 80 % respectively for average influent nitrogen and phosphorus concentration of nearly 39.63 mg/L and 11 mg/L. At this steady state bio film state the performance for ammonium removal was about 98 % at influent concentration of 22.54 mg/L. To conclude from the present investigation, in addition to excellent organic removal, significant reduction of nutrients could also be achieved by the trickling filter in the treatment of brewery wastewater.

Key words: Bioreactor, performance evaluation, brewery wastewater, flow rates, fixed bed bio film process, phosphorous, nitrogen

5.1 Introduction

The two primary nutrients of concern are phosphorus and nitrogen (Rob Leeds et al., 2006). When an ecosystem experiences an increase in nutrients, primary producers reap the benefits first. In aquatic ecosystems, species such as algae experience a population increase (called an algal bloom). Algal blooms limit the sunlight available to bottom-dwelling organisms and cause wide swings in the amount of dissolved oxygen in the water. Oxygen is required by all aerobically respiring plants and animals and it is replenished in daylight by photosynthesizing plants and algae. Under eutrophic conditions, dissolved oxygen greatly increases during the day, but is greatly reduced after dark by the respiring algae and by microorganisms that feed on the increasing mass of dead algae. When dissolved oxygen levels decline to hypoxic levels, fish and other marine animals suffocate. As a result, creatures such as fish, shrimp, and especially immobile bottom dwellers die off (Horrigan, 2002). In extreme cases, anaerobic conditions ensue; promoting growth of bacteria such as *Clostridium botulinum* that produces toxins deadly to birds and mammals, dead zones.

Policy changes to control point sources of phosphorus have resulted in rapid control of eutrophication. The World Resources Institute has identified 375 hypoxic coastal zones in the world, concentrated in coastal areas in Western Europe, the Eastern and Southern coasts of the US, and East Asia, particularly Japan (Selman, 2007). Therefore in order to minimize the environmental problems associated with nitrogen and phosphorous in wastewater it is imperative to find ways and means of decreasing it before discharging nutrient rich wastewater into surface waters.

The current wastewater treatment methods for the removal of nitrogen and phosphorous can be divided into three categories which are physical, chemical and biological wastewater treatment processes. The physical units most commonly used in wastewater treatment include screening, grit removal, mixing and flocculation, sedimentation, clarification, aeration, and volatilization and stripping of volatile organic compounds (VOCs) (Nemerow and Agardy, 1998; Tanyi, 2006). These methods only remove 20 % to 30 % of phosphorus in wastewater (Henze *et al.*, 1995) and are mostly used at pre-treatment stage (Gillberg *et al.*, 2003). The widely used chemical wastewater treatment method includes chemical coagulation, precipitation, disinfection and oxidation, ion exchange and others (Henze *et al.*, 1995; Gillberg *et al.*, 2003). Using precipitation up to 90 % of total phosphorus, 25 % of total nitrogen, 90 % of BOD and 90 % of suspended solids can be removed (Gillberg *et al.*, 2003). This type of treatment mainly relies on addition of chemicals and is applied when the wastewater cannot be treated biologically (Tanyi, 2006).

A significant disadvantage of this treatment process is additive processes involve (Gillberg *et al.*, 2003; Tanyi, 2006). As a result there is a net increase in the dissolved constituent in the wastewater. Besides that another disadvantage of chemical treatment process is that the cost of most chemicals is related to the cost of energy (Gillberg *et al.*, 2003). Therefore, the end user has little control over chemical costs. Chemical precipitation of phosphorous primarily uses aluminum and iron coagulants or lime to form chemical flocs with phosphorus (Gillberg *et al.*, 2003). These flocs are then settled out to remove phosphorus from the wastewater. However, compared to biological removal of phosphorus, chemical processes have higher operating costs, produce more sludge, and result in added chemicals in sludge (Gillberg *et al.*, 2003; Metcalf and Eddy, 2003; Tanyi, 2006).

As biological phosphorous removal does not require chemical precipitants and produces less waste sludge, it forms a good alternative to chemical phosphorous removal in wastewater treatment plant (Liu *et al.*, 2007). Phosphorus and nitrogen removal from wastewater is an important strategy to control eutrophication, and biological nutrient removal (BNR) is an effective and economical way to remove phosphorous along with nitrogen and organic materials from wastewater (Ouyang *et al.*, 1999; Erdal *et al.*, 2000; Yong Ma *et al.*, 2005; Liu *et al.*, 2007). Industrial effluent total phosphorus comprises soluble and particulate phosphorus. Particulate phosphorus can be removed from wastewater through solids removal. To achieve low effluent concentrations, the soluble fraction of phosphorus must also be targeted (US EPA, 2005). Conventional secondary biological treatment systems accomplish phosphorus removal by using phosphorus for biomass synthesis during BOD removal. A typical phosphorus content of microbial solids is 1.5 - 2 % based on dry weight. Wasting of excess biological solids with this phosphorus content may result in a total phosphorus removal of 10 - 30 %, depending on the BOD-to-phosphorus ratio, the system sludge age, sludge handling technique, and side stream return flows (US EPA, 1987; Cretu and Tobolcha, 2005). More phosphorous can be removed if one of a number of especially developed biological phosphorous removal process is used. These processes are based on the exposure of microbes in an activated-sludge system to alternating anaerobic and aerobic conditions. This stresses the micro-organisms, so that their uptake of phosphorous exceeds normal levels (Erdal *et al.*, 2000; Metcalf and Eddy, 2003).

Industrial effluent total nitrogen comprises ammonia, nitrate, particulate organic nitrogen, and soluble organic nitrogen (Jeyanayagam, 2005). The biological processes that primarily remove nitrogen are nitrification and denitrification (Seyoum Leta, 2004; Jeyanayagam, 2005). Nitrification occurs in the presence of oxygen under aerobic conditions, and denitrification occurs in the absence of oxygen under anoxic conditions (US EPA, 2005). During nitrification ammonia is oxidized to nitrite by one group of autotrophic bacteria, most commonly *Nitrosomonas* (Surampalli *et al.*, 1997; Metcalf and Eddy, 2003; Seyoum Leta, 2004). Nitrite is then oxidized to nitrate by another autotrophic bacteria group, the most common being *Nitrobacter*. Following nitrification the next process is denitrification which involves the biological reduction of nitrate to nitric oxide, nitrous oxide and nitrogen gas (Surampalli *et al.*, 1997; Metcalf and Eddy, 2003; Seyoum Leta, 2004). Following these processes nitrogen can be removed from the wastewater. The process of nitrogen removal by bacterial conversions in trickling filter follows a series of reactions as in a nitrogen cycle (Figure 5.1).

Anaerobic treatment systems include lagoons, continuous stirred tank reactors (CSTR), anaerobic filters (AF), up flow anaerobic sludge blanket (UASB), fluidised bed (FB), expanded granular sludge bed (EGSB) and internal circulation (IC) reactors (Driessen and Vereijken, 2003). Several types of anaerobic reactors can be applied to brewery wastewater treatment; however, the UASB reactor is the world's most widely applied anaerobic reactor system for treatment of brewery effluent (Batson *et al.*, 2004; Parawira *et al.*, 2005).

Anaerobic processes have been used successfully for many years for highly polluted wastewaters from sugar factories, distilleries, wood pulp factories, etc and have also become of more interest to other sectors of industry, such as breweries, since energy in the form of combustible methane gas can be recovered (Kunze, 2004; Brito et al., 2005). Driessen and Vereijken (2003) reported that this treatment process have 70%-85% COD and low nutrients (N and P) removal efficiency. Stadlbauer *et al.* (1994) reported COD removal efficiencies of 85% to 90% from a study of anaerobic purification of lager beer brewery wastewater in a laboratory scale bio film reactors with and without a methanation cascade. Austermann-Haun and Seyfried (1994) also reported 80 % COD removal efficiency from the pilot-scale UASB reactor treating clear beer brewery wastewater. Further a study conducted by Brito *et al.* (2005) also indicated that 70% to 80 % COD removal by the UASB process in Unicer SA brewing industry. However, in spite of such efficiency the NH_4^+-N levels were above the threshold values prescribed for wastewater discharge in surface waters (Brito *et al.*, 2005). The effluent of an anaerobic digester contains high amounts of nutrients, most in the form of ammonia and also phosphorous (Obaja *et al.* 2003; La Motta *et al.*, 2007). Thus, anaerobic biological treatment alone cannot achieve the performance levels required for direct discharge in to receiving streams. Never the less, it can be employed as a cost effective pretreatment ahead of aerobic treatment (La Motta *et al.*, 2007).

Aerobic activated sludge treatment is the most frequent and widely applied treatment technology. According to Driessen and Vereijken (2003) aerobic systems can achieve 90% - 98 % COD and high nutrients (N/P) removal. In an aerobic process, typical concentrations of phosphorus in heterotrophic microorganisms are between 10 to 25 g/kg VSS and 7 to 18 g/kg COD (Henze *et al.*, 1995). Aerobic treatment processes have traditionally been employed for reduction of BOD, but concurrent reductions of other contaminants often proves infeasible without coupling aerobic treatment with anaerobic or anoxic pretreatment (Al-Rekabi1 *et al.*, 2007).

In conventional aerobic processes like activated sludge process due to the oxidative biological reaction, large amounts of biomass are produced which settle as sludge which requires further disposal (Driessen and Vereijken, 2003; Kanagachandran and Jayaratne, 2006). The aerobic treatment of brewery effluent requires a comparatively large energy input compared to anaerobic treatment (Kanagachandran and Jayaratne, 2006).

In anaerobic-aerobic treatment of wastewater the main objectives are to enhance organic matter removal as well as to promote the removal of components which are barely affected by the anaerobic treatment i.e. nutrients and pathogens (Chernicharo and Nascimento, 2001; Driessen and Vereijken, 2003). The fact that the ability of anaerobic treatment systems to biologically transform toxic organic chemicals to forms more easily degraded in aerobic environments greatly expands the capabilities of biological systems for treatment of organic wastes (Al-Rekabi1 *et al.*, 2007).

Activated sludge process is the most common feature in most of the studies along with various pre-treatment options. However, single and two stages activated sludge processes are reported to be inefficient with inadequate nitrification (Luthy, 1981; Pandey *et al.*, 1991). Two-and three-stage anoxic-aerobic/anaerobic-anoxic-aerobic sequential treatment enhanced process performance (Wen *et al.*, 1991; Zitomer and Speece, 1993; Zhang *et al*., 1997; Li *et al.*, 2003).

Typically, a BNR process has anaerobic, anoxic and aerobic (A/A/A) reactors; the activated sludge is exposed to these conditions repeatedly. In the anaerobic reactor the activated sludge releases phosphorus and accumulates polyhydroxyalkanoates (PHA) when the carbon substrate is more abundant (Jenkins and Tandoi, 1991; Ouyang *et al.*, 1999; Erdal *et al.*, 2000; Wilderer *et al.*, 2001; WEF and ASCE/EWRI, 2006). During the subsequent aerobic phase, polyphosphate accumulating organisms (PAOs) take up phosphate from the bulk liquid and store it in the form of polyphosphate, while PHA is used as a carbon and energy source (Jenkins and Tandoi, 1991; Ouyang *et al.*, 1999; Wilderer *et al.*, 2001; Pijuan *et al.*, 2005; Harper *et al.*, 2006; Mullan *et al.*, 2006), therefore, phosphorus can be removed from wastewater. In the anoxic reactor, nitrate decreases due to denitrification (Ouyang *et al.*, 1999).

Besides its use for the removal of nitrate, anoxic reactor can also be used for biological phosphorus removal (Tanyi, 2006). Many researchers have shown that some PAOs use nitrate instead of free oxygen to oxidize stored PHAs and take up phosphorus. These denitrifying PAOs remove phosphorus in the anoxic zone, rather than in the aerobic zone (Mino *et al.*, 1995; Barker and Dold, 1996; Ng *et al.*, 2001; Jeyanayagam, 2005). Moreover in the aerobic reactor, nitrification, organic substrate oxidation, and phosphorus uptake occur at the same time (Ouyang *et al.*, 1999; Peng *et al.*, 2006). Consequently, organic substrate, nitrogen and phosphorus can be removed simultaneously in the process. However such wastewater treatments obviously need extra cost for the special arrangements, high operation skill, and large foot print. Therefore specially might not be suitable for technologically less developed countries.

In fixed film bioprocess like trickling filters relatively low power requirements; they require power for pumping only and do not need large power-hungry aeration blowers. From motor-driven rotary distributors are powered by fractional horsepower electric motors. Moreover they produce less sludge than suspended-growth systems. In addition the sludge tends to settle well because it is compact and heavy. Therefore the employment of trickling filter which has the advantage of having low cost, without very complex arrangements, low operational skills and small foot print is obviously a better alternative for the reduction of nitrogen and phosphorous contained in brewery wastewater.

The bioreactor considered here was originally constructed and designed for reduction of both organics and nutrients simultaneously from brewery wastewater. There was not any special arrangement made on the trickling filter for enhancement of any one of the nutrients removal. However as it can also be seen from the discussion here under besides excellent efficiency of organics removal the trickling filter attained significant reduction of nitrogen and phosphorous at the given operating condition without extra cost for special designing and operation of the bioreactor.

In the present research the hypothesis was that in a trickling filter since anaerobic, anoxic and oxic bio film zones could be developed and therefore, advantage of simultaneous organic and nutrient removal that is achieved in (A/A/A) could be achieved in the trickling filter.

The objective of this chapter was therefore to evaluate the performance of a mineral filled trickling filter in reducing the concentration of nitrogen and phosphorous during the treatment of brewery wastewater.

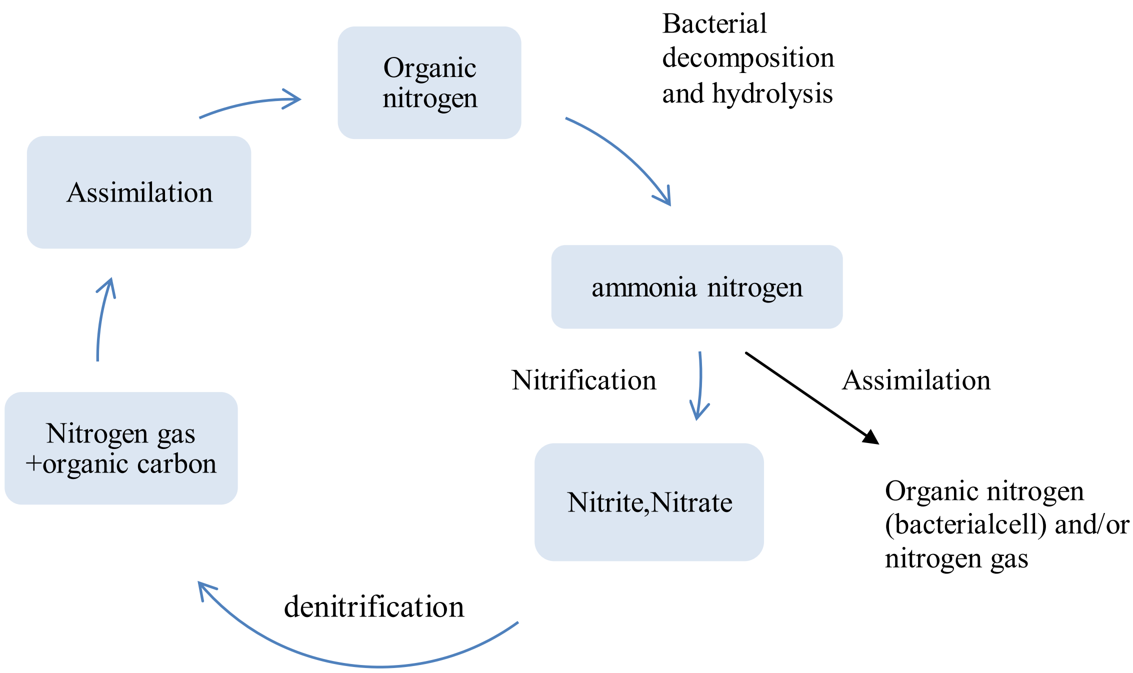


Figure 5.1 Nitrogen transformation in biological treatment processes

5.2 Result and discussion

5.2.1 Influence of hydraulic loading rates

As can be seen from Fig. 5.2 below there was no special trend for both nitrogen and phosphorus removal with increase in hydraulic loading rate. This might be due to absence of any special arrangement made on the trickling filter for established removal of both nitrogen and phosphorus. Enhancement of aeration at higher hydraulic load can facilitate nitrification. Therefore denitrification could follow using the nitrified trickling filter effluent. On the other hand enhancement of aeration has negative impact on denitrification there by total nitrogen removal performance of the trickling filter vary.

But the trickling filter tends to achieve higher efficiencies for both nitrogen and phosphorus at higher hydraulic loading rates. Here also this is due to the fact that the bio film thickness control at the higher hydraulic load will reduce the internal diffusion limitation there by enhancing substrate transfer (P.W. Westerman *et al.,* 2000; Vayenas, D.V., *et al.*,1997).

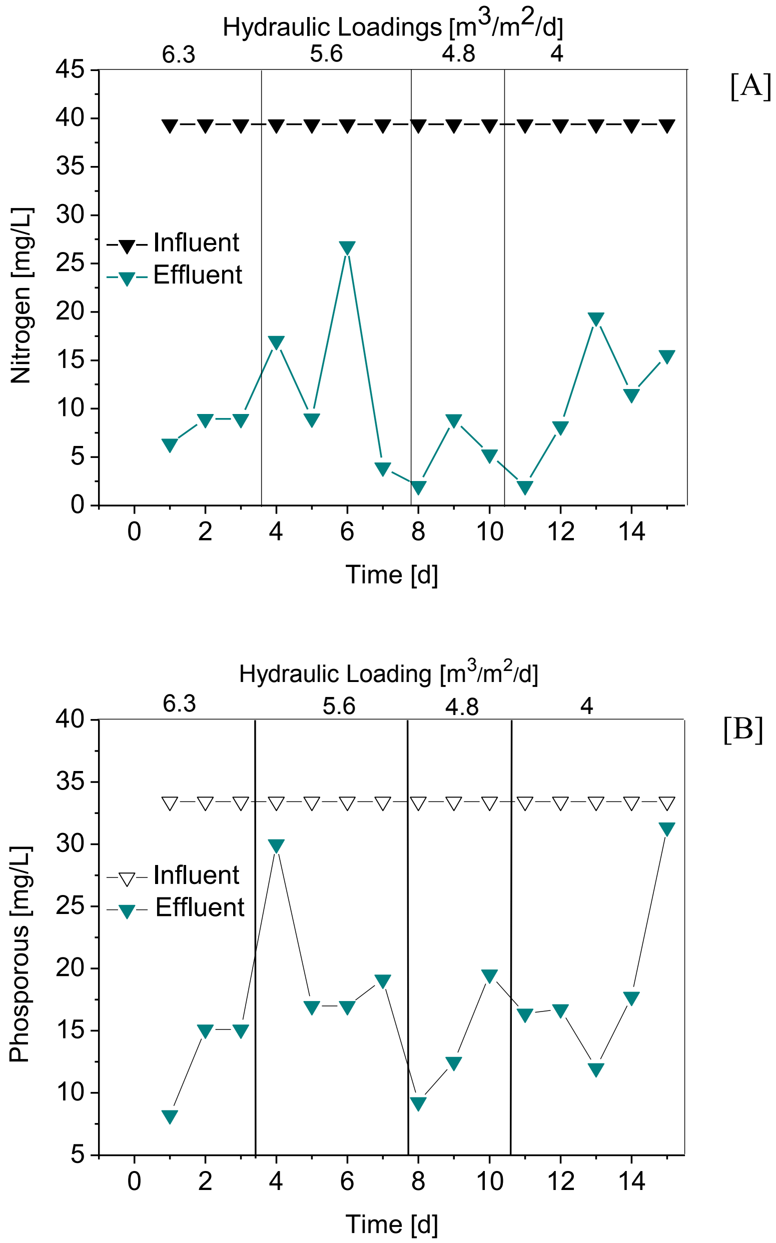


Figure 5.2 The trend in nitrogen and phosphorus reduction as a function of hydraulic loading rate , [A] & [B]

The primary purpose of this investigation was to evaluate the response of the trickling filter with respect to nitrogen and phosphorus removal when the wastewater flow changes. Total nitrogen and total phosphorus removal effeciencies at the maximum value of hydraulic loading rate i.e. at 6.3 $m^3/m^2/d$ was 75.3 ± 2.27% and 71.80 ± 8.07 % respectively. The performance of the trickling filter with the change in hydraulic loading is illustrated in Fig. 5.3. Therefore this investigation confirms the absence of any detrimental effect on the performance of the trickling filter with respect to nitrogen and phosphorus removal due to hydraulic shock loads during the treatment of brewery wastewater. The raw data table for this figure is given in Annex A5

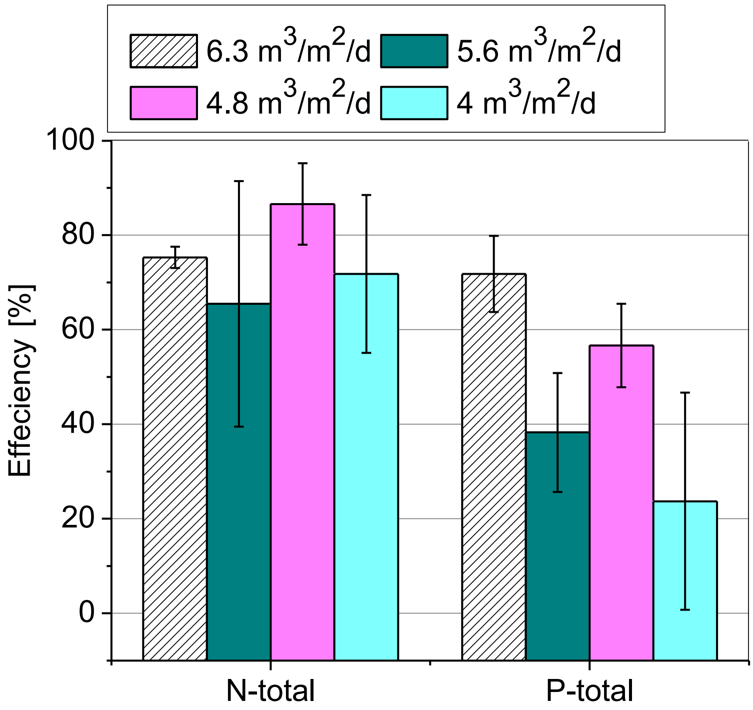


Figure 5.3 Performance of the trickling filter at different hydraulic loading rate

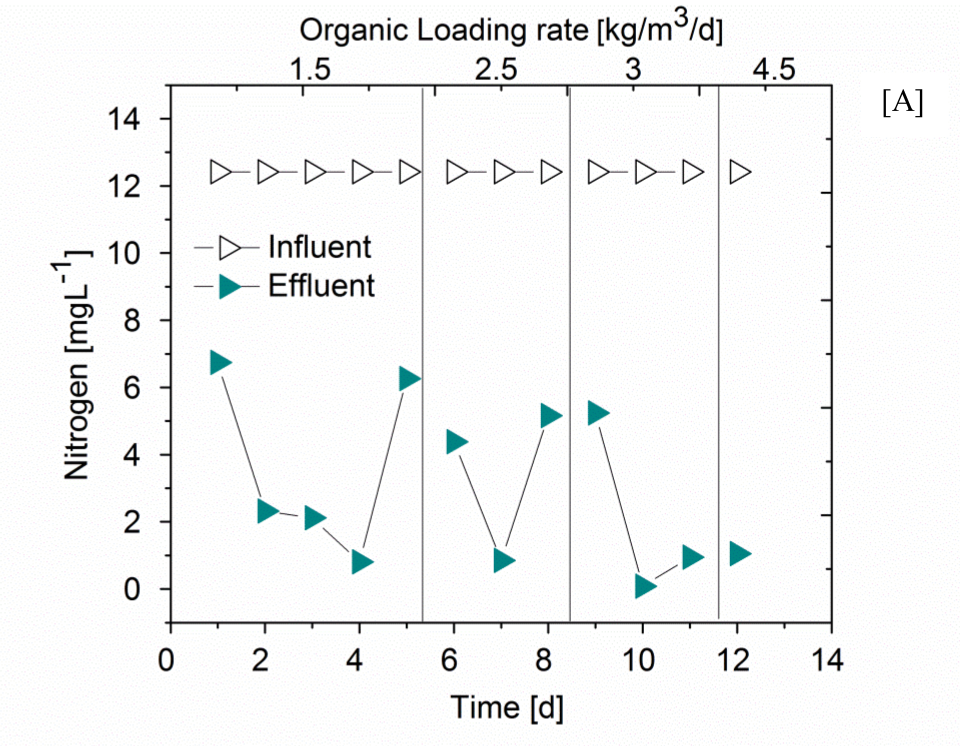
5.2.2 Influence of COD load

The response of the trickling filter to different initial influent COD concentration was also investigated. Four different influent COD concentrations were chosen namely 1160 mg/L, 1648 mg/L, 2325 mg/L and 3070 mg/L. Like in the finding of hydraulic loading rates, the removal efficiency for both nitrogen and phosphorus increased with influent COD concentration in general (Figure5.4). But the trickling filter had minimum efficiency for phosphorus at 2323 mg/L, phosphorus removal 59.51%. An average nitrogen and phosphorus removal of 93.07 ± 2% and 72.45 ± 8% respectively was recorded at the maximum influent COD concentration. However at this high COD concentration the trickling filter tends to develop excess biomass rapidly (in about 18 days) and the effluent COD concentration was high as also indicated in the previous chapter. Therefore this COD loading was not considered as the design COD loading of the trickling filter. The optimum COD loading is 1163 mg/L at which the efficiency of removal for total nitrogen and total phosphorus were 70.6 ± 21.5% and 68.87 ± 8.52 % respectively. The raw data table for this figure is presented in Annex B4.

At high organic loadings the hetrotrophic bacteria inside the zooglea will be favored for the conversion of nitrate to nitrogen gas in the anoxic zone of the trickling filter. In the anoxic zone of the trickling filter nitrate will be used as electron acceptor when sufficiently high carbon source is available. The high phosphorus removal in the trickling filter could suggest some how suitability of the organic substances of the wastewater to be utilized as a carbon source by the $PAO_{s,}$ therefore the uptake of phosphorus in excess of the normal metabolic requirment could take place in the aerobic zone of the trickling filter. As a result phosphorus uptake and nitrification could take place in the aerobic zone of the trickling filter. In the anoxic zone the action of $DPAO_s$ (denitrifying phosphate accumulating organisms) remove at the same time nitrate and phosphorus. The average efficiency of nitrogen and phosphorus removal at the different initial wastewater concentration is as shown in Fig. 5.5. And the raw data table is given in Annex B6

In Fig. 5.6 plot of the removal rate versus loading rate as a function of hydraulic loading rate with respect to total nitrogen and total phosphorus removal by the reactor is shown. According to the results, the removal rate showed a strong correlation with the loading rate. The removal efficiency for total nitrogen ranged from 71.8 % to 86.59 % as the HLRs increased from 400 L/m^2/d to 800 L/m^2/d, and it was ranged from 65.46 % to 75.3 % as the HLRs changed from 1600 L/m^2/d to 2400 L /m^2 /d at average nitrogen influent concentration of 36.9 mg/L. The mass removal rate for TN was ranged from 9.20 g/m^2/d to 68.57 g/m^2/d, influenced moderately by increasing MLR. Therefore the change in efficiency of the trickling filter for total nitrogen removal has shown only small difference as the HLRs changes from 400 L/m^2/d to 2400 L/m^2/d.

Mass removal rate for total phosphorus in this study ranged from 1.34 g/m^2/d to 12.06 g/m^2/d as the mass loading rate increased from 12.20 g/m^2/d to 24.41 g/m^2/d, influenced significantly by increasing MLR and it was ranged from 19.25 g/m^2/d to 42.48 g/m^2/d as the mass loading rate changed from 48.82 g/m^2/d to 73.22 g/m^2/d. As a result the trend of efficiency for phosphorus removal increased as the HLRs changed from 400 L/m^2/d to 800 L/m^2/d and almost no change as HLRs changes from 1600 L/m^2/d to 2400 L/m^2/d.



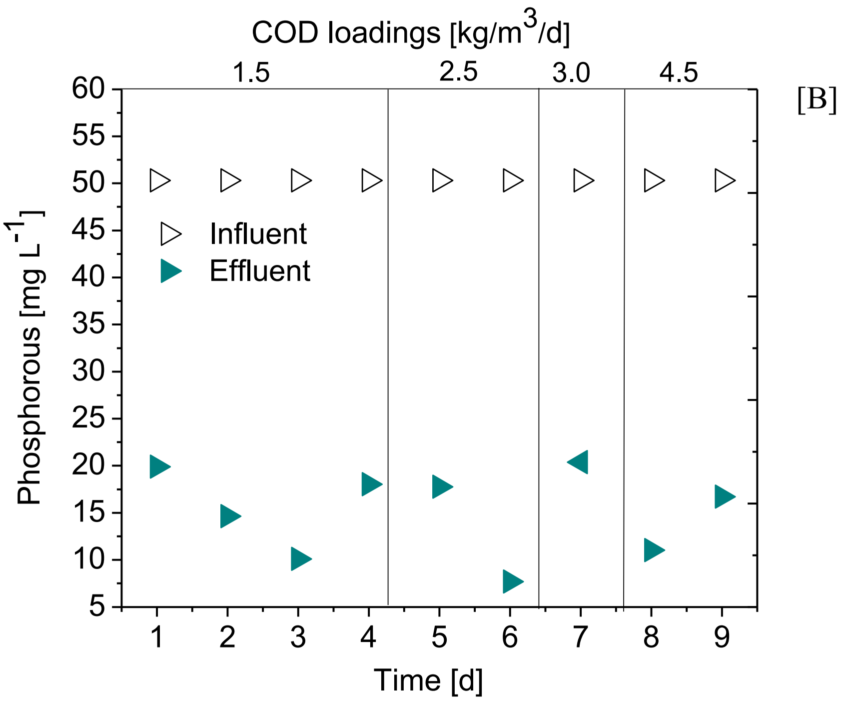


Figure 5.4 **The trend in nitrogen and phosphorus reduction as a function of organic loading rate, [A] & [B]**

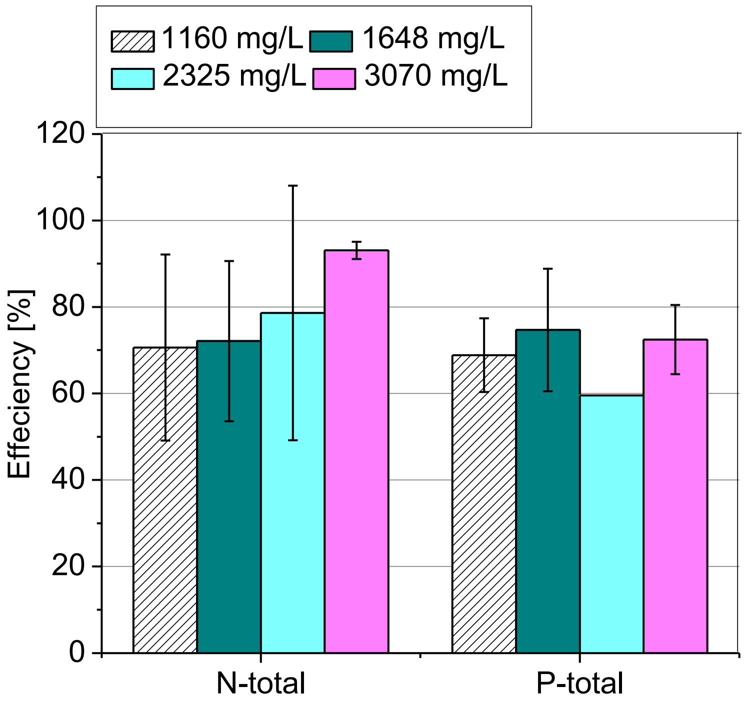
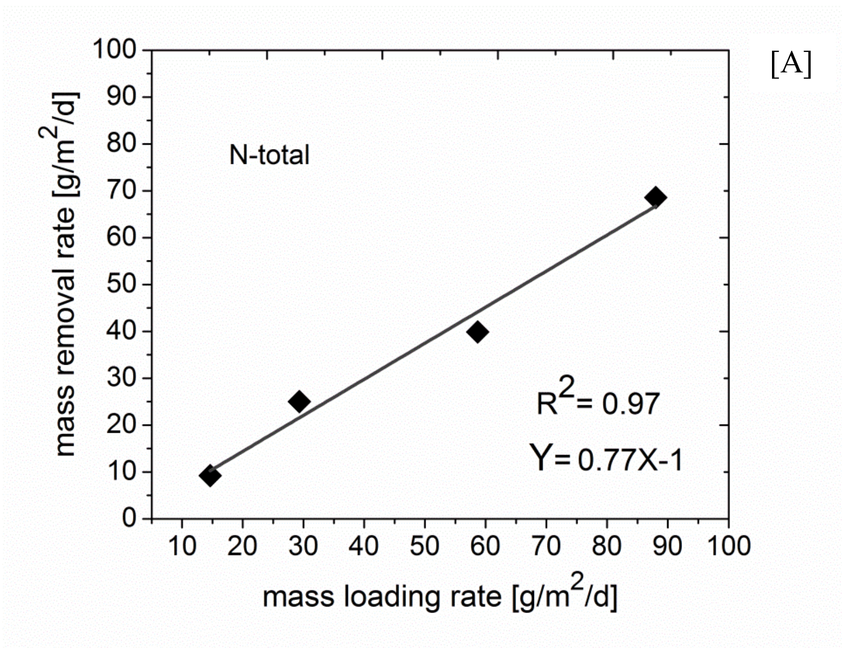


Figure 5.5 Performance of the trickling filter at different COD loading



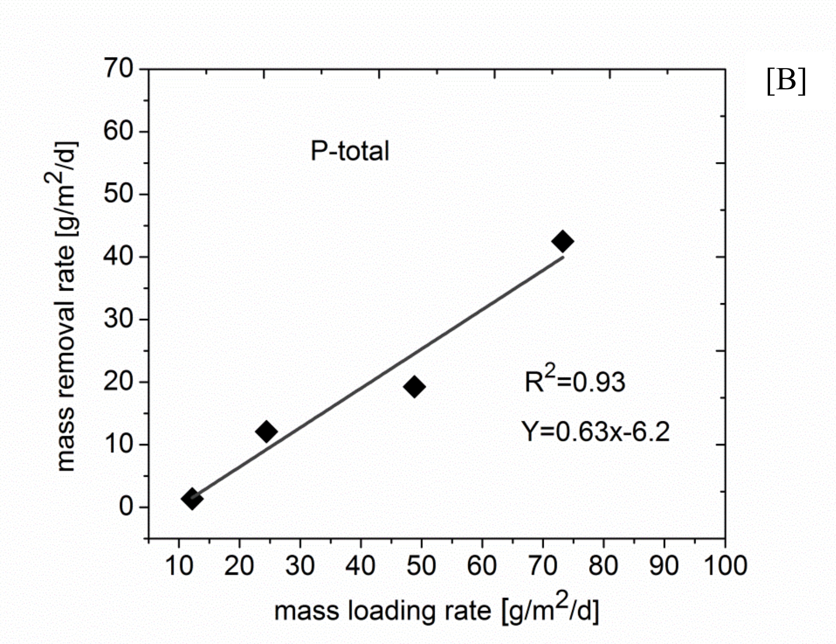


Figure 5.6 the correlation of mass loading rate and mass removal rate as a function of hydraulic loading in the removal of nitrogen and phosphorus, [A] & [B]

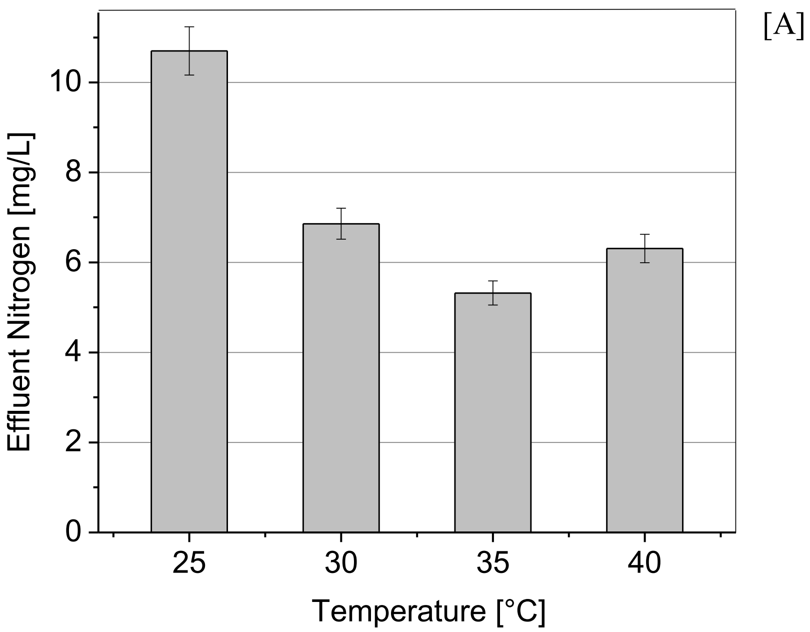
5.2.3 Influence of temperature and pH on the trickling filter performance

To investigate the effect of temperature on nutrient removal performance of the trickling filter, temperature of the feed wastewater was adjusted to five different temperatures namely 25°C, 30°C, 35°C and 40°C which is typical temperature range of most breweries. The wastewater kept at each temperature for three consecutive days and the average performance is reported in all cases. Total nitrogen removal rate at 25°C was 9.08 $g/m^2/d$ and it was increased to values of 18.25 $g/m^2/d$ and 21.93 $g/m^2/d$ as the temperature increased from 25°C to 35°C. And the removal rate dropped from 21.93 to 19.56 [g $/m^2/d$] as the temperature increased from 35°C to 40°C, indicating the optimum temperature is 35°C for the given range. A more gradual but similar trend of removal with temperature was observed during the removal of phosphorous, but the optimum temperature was 30°C in case of phosphorus. Fig.5.7 illustrates the trend of concentration reduction at the different temperatures.

The performance of the trickling filter for the selected parameters as a function of different wastewater pH was also investigated in this study. With regard to the removal of ammonium nitrogen and total phosphorus the trickling filter achieved maximum removal efficiency at near neutral pH. As the pH increased from 5.65 to 6.3 the removal efficiency for ammonium nitrogen increased from 55.5 % to 95.6 % and for that of total phosphorus it increased from 36 % to 70 %. The total nitrogen removal achieved at pH 8.3 was 82.73 % compared to 10.01 % at a pH of 9. The response of the trickling filter for change in pH was also the same during the removal of COD (see chapter three), that implies high removal efficiency for all COD, nitrogen and phosphorus can be achieved at the same pH condition which makes feasible the field scale application of the trickling filter. The trend of effluent concentration as a function of pH is depicted in Fig. 5.8.

5.2.4 Influence of ratio of wastewater organic content to nutrient

The purpose of conducting this experiment is to evaluate the causes and effects of unfavourable nutrient ratios, and to analyze the measures to be taken to deal with them. The content of the individual nutrients in wastewater should correspond to the needs of the bacteria in the trickling filter sludge, and there should be a balanced relationship between C, N and P. This is crucial to the effectiveness of the biodegradation processes. Therefore operation of the trickling filter at different C: N: P ratio was carried out while keeping all the other operating conditions constant. A flow rate of 300 L/d, temperature varying at 20°C and pH varying at 7 was the operating condition. The variation in total nitrogen removal efficiency with the different ratio was significant where as in case of total phosphorus removal there was only slight variation. The optimum C: N: P ratio was 62:4:1 for this particular investigation at which the removal (%) of COD, nitrogen and phosphorus was 85.5%, 61.87% and 37.5% respectively. Fig.5.9 depicts the removal efficiencies by the trickling filter at the different carbon to nutrient ratio.



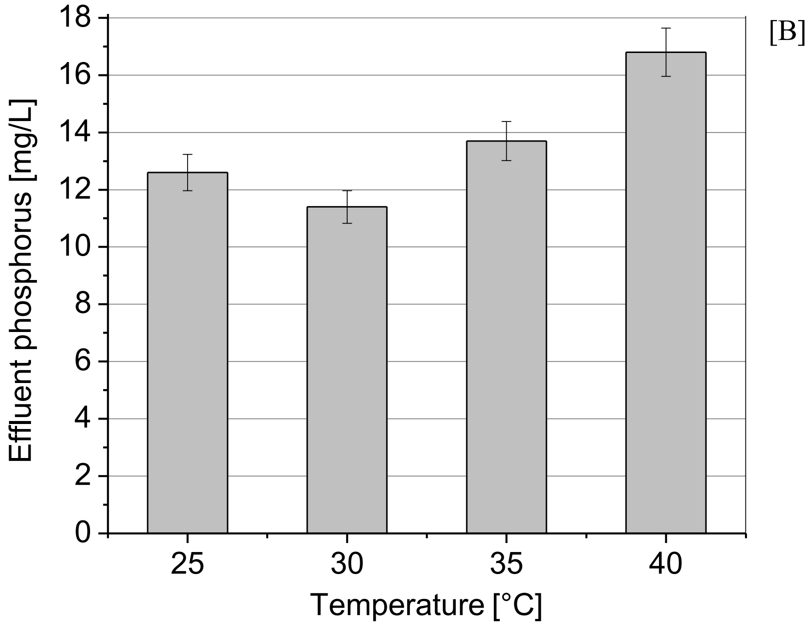
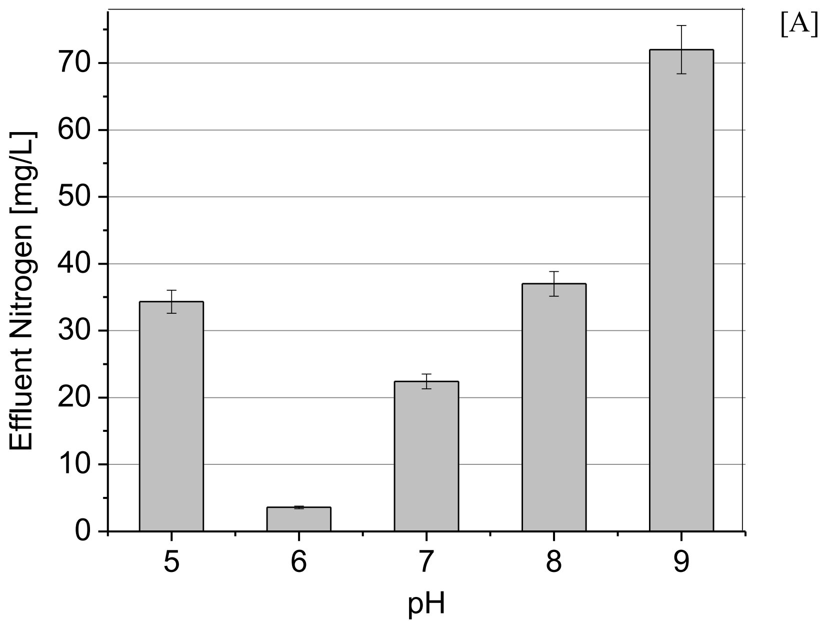


Figure 5.7 **Trend in the removal of nitrogen and phosphorus at the different temperature ,[A] & [B]. Influent nitrogen=14.5 mg/L and influent phosphorus=17.5 mg/L**



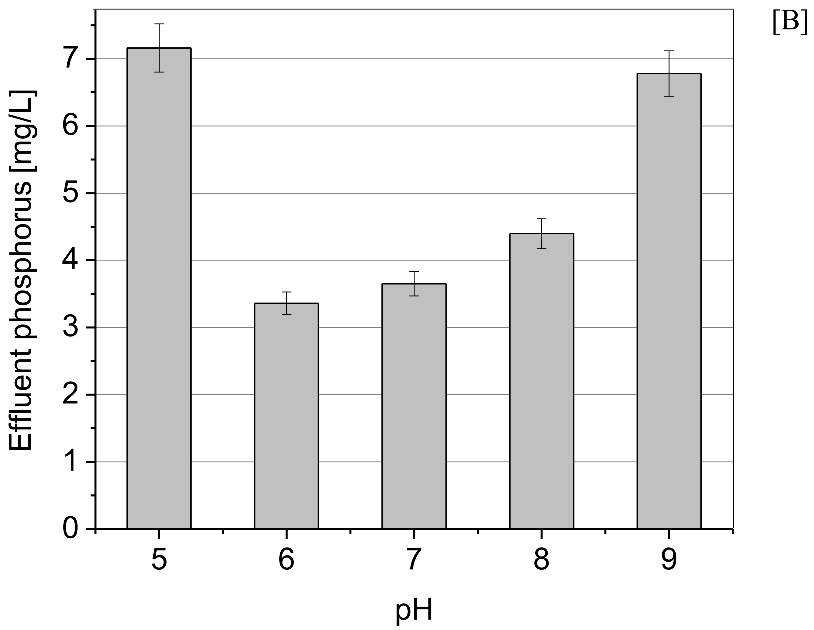


Figure 5.8 **Trend in the removal of nitrogen and phosphorus at the different pH, [A] & [B] with feed Nitrogen 80 mg/L & Phosphorus 14 mg/L**

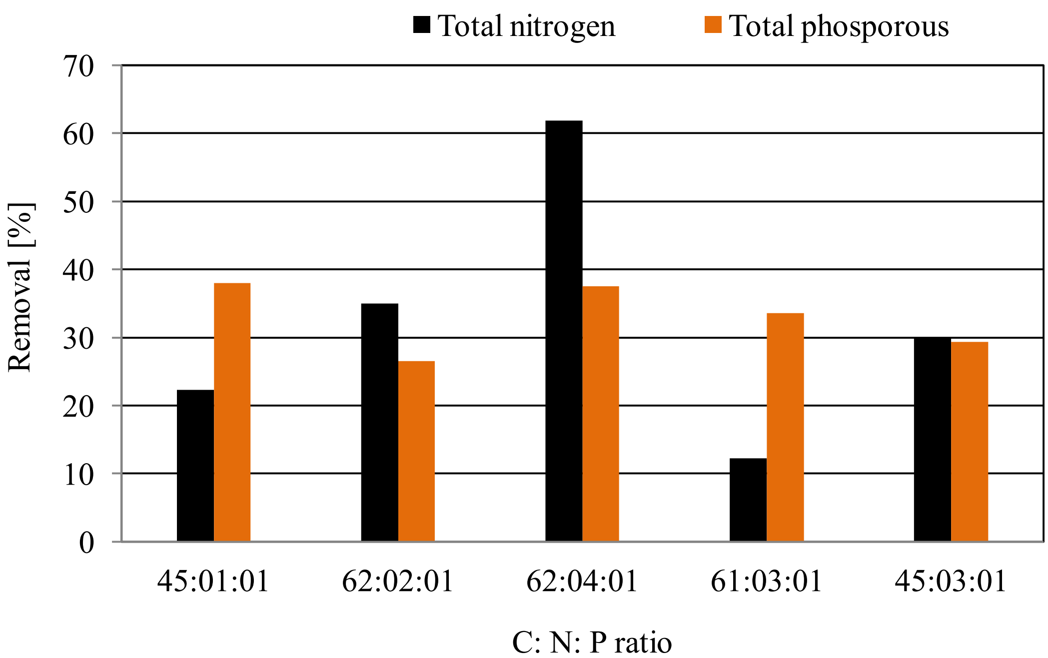


Figure 5.9 **The removal of total nitrogen and total phosphorus as a function of C: N: P ratio**

Performance at steady state bio film state of the trickling filter was evaluated for each selected parameters at constant operating conditions. Fig 5.10 illustrates the influent and effluent nitrite and nitrate concentration after the trickling filter achieved steady state with respect to COD removal. The absence of high nitrite or nitrate level in the effluent some how indicates the occurence of denitrification by the bioreactor. And the trend of removal at constant COD concentration (nearly 1000 mg/L) and constant wastewater inflow (300 L /d) during this operation for NH_4N, total nitrogen and total phosphorus is given in Fig.5.11 [A] to [C]. Where as Fig.5.12 portraits the statistical description of the trickling filter performance for nitrogen, phosphorus and ammonium nitrogen removal. It was difficult in general to achieve steady state with respect to the removal of both nitrogen and phosphorus. The trend of removal given in the figures was obtained when the trickling filter achieved steady state only with respect to removal of organics removal. At influent total nitrogen of around 39.63 mg/L, total effluent nitrogen level in the range of 2.9 mg/L−27.8 mg/L was achieved. The nitrogen removal efficiency was only slightly varying at 88 %. The high volume of packed filter medium (about 0.2 m^3) for the given loading condition ,which is large enough for the accommodation of both the authotrophs and hetrotrophs could be one reason for the unusually good nutrient removal performace of the trickling filter here. The prevailing mechanism of nitrogen removal by the trickling filter is discussed here under.

The basic principle behind the removal of nitrogen by the trickling filter is similar with what is conventionally applied in suspended solids growth systems, where nitrified mixed effluent is returned to an anoxic zone where denitrification occurs. In the present operation of the trickling filter recirculation of TF effluent introduces nitrate into the top of the filter where heterotrophic activity, and therefore potential denitrification activity, is highest. This process may remove up to 50 % of the trickling filter's NO_3^- when a 100 % recycle rate is used and up to 67 % of the trickling filter's nitrate when a 200 % recycle rate is used (Metcalf and Eddy, 2003). This makes for an attractive alternative TN reduction upgrade where discharge limits are relatively high and only partial denitrification is required. In trickling filter process total nitrogen (TN) reduction ranging from 0 % to over 50 % can be achieved across the TF process (Pearce , 2004). Similar TF nitrogen removal performance was reported from the pilot work conducted at the Littleton/Englewood wastewater Treatment Plant (WWTP) in USA (Biesterfeld et al., 2003) and this has been explored by other researchers (Dorias and Baumann, 1994; Vanhooren et al., 2003).

The incorporation of recirculation scheme in the trickling filter however has a potentially negative effect on denitrification as it increases the wetting process, and hence improves the oxygen transfer through the filter. The potential of the anoxic mechanisms in the trickling filter will be reduced by any improved dissolved oxygen penetration into the biofilm (Pearce, 2004). Whilst the percentage nitrogen removal will increase with COD load due to a higher oxygen demand at the top of the filter, nitrogen removal will reduce at higher COD loads where nitrification is suppressed. Because there is a conflict of requirements for simultaneous nitrification and denitrification it appears that the achievable reliable process performance through process control is limited to 30-50 % nitrogen removal (Pearce, 2004).

In the present trickling filter total nitrogen removal as high as 88% was achieved including the bacterial assimilation.

Where as for total phosphorus, effluent value in the range of 1.8 mg/ L −7.16 mg/ L was achieved for an inlet concentration of 10.92 mg/L and when the trickling filter achieved steady state with respect to organic removal the efficiency of removal for total phosphorus was about 80 %. Influent ammonium nitrogen varied at about 22.54 mg/L and effluent ammonium nitrogen value in the range of 0.2 mg/L−13.6 mg/L was achieved. At longer operation time of the trickling filter most probably due to the increase in population of the slow growing nitrifer, a near to 100 % removal efficiency of ammonium nitrogen was achieved at this steady state. The raw data tables for all the figures just discussed above are given in annexes A7 to A11.

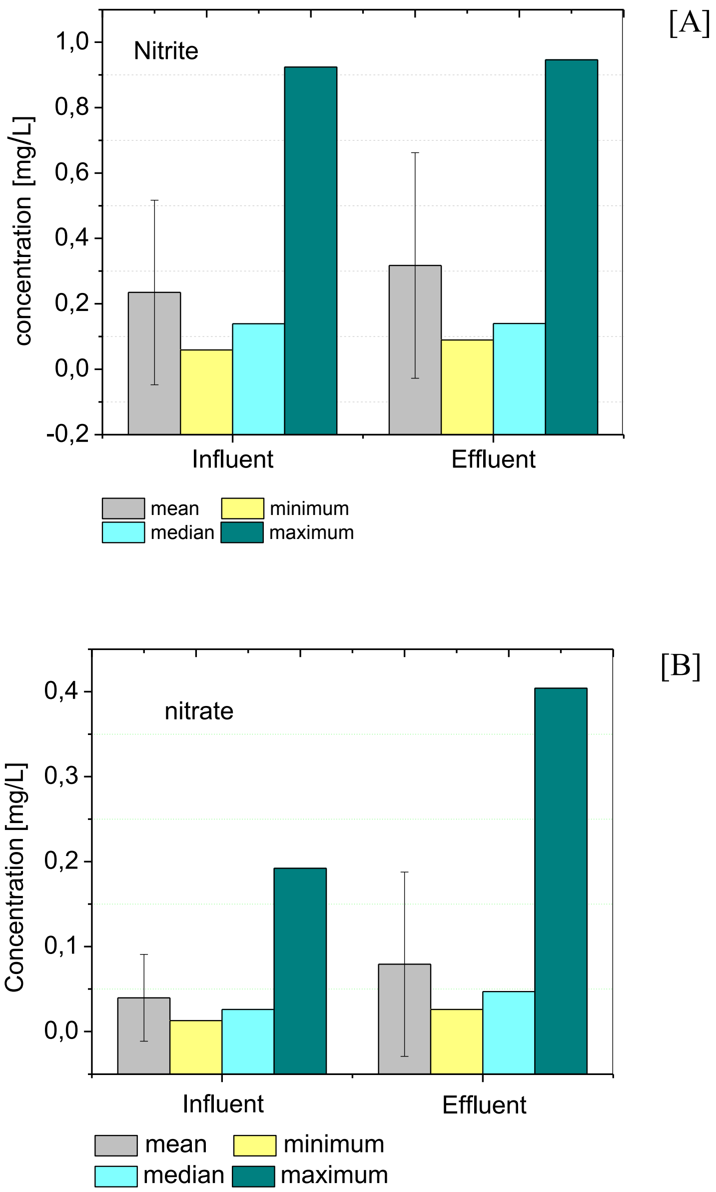
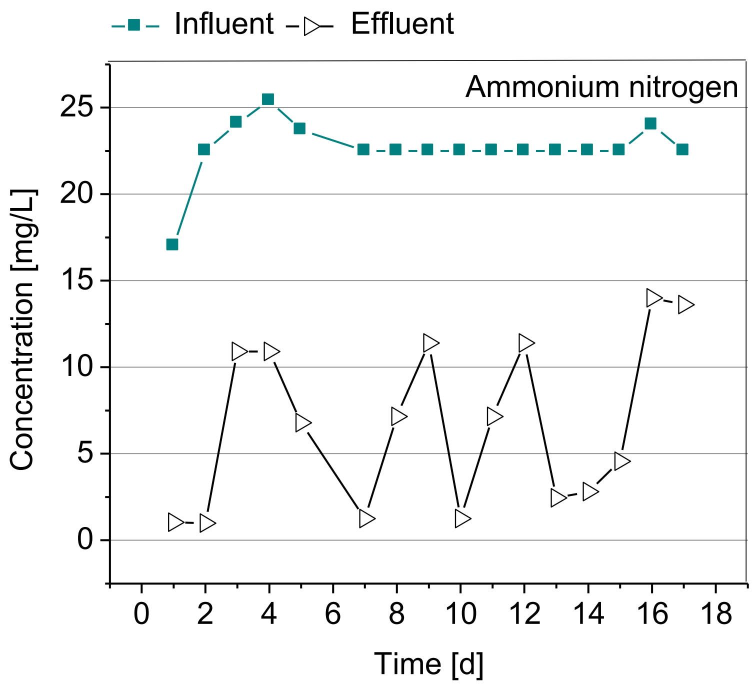


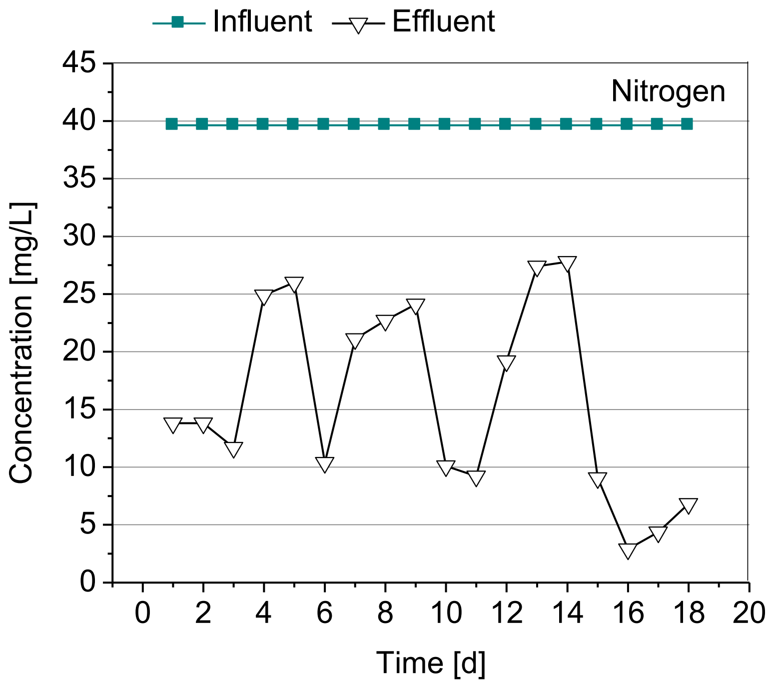
Figure 5.10 Profile of nitrite and nitrate in the trickling filter, [A] & [B]

The variation in performance from the present study may be explained by the presence of communities of quite different compositions developing in the trickling filter even if the trickling filter is kept at constant operation conditions. Glycogen accumulating organisms, phosphate accumulating organisms and ammonia oxidizing bacterium are influenced by change in operating conditions such as O_2, type and amount of substrate, temperature and pH. However, during this operation of the trickling filter all the operating condions were kept at constant. Therefore the most probable reason for the variation in the removal of phosphorus and nitrogen is the change in biofilm thickness and therefore microbial composition with time.

In the removal of nitrogen as the bio film gets ticker most of the organic matter will be degraded near the surface of the bio film and the suppression of nitrification would not occur near the lower section of the trickling filter. In the removal of phosphorus the growth of DPAOs could take place as the biofilm gets ticker. In general change in the microbial community structure along the depth of the trickling filter as the biofilm thickness changes is belived to be the major cause for the variation of nitrogen and phosphorus removal at the steady state.



[A]



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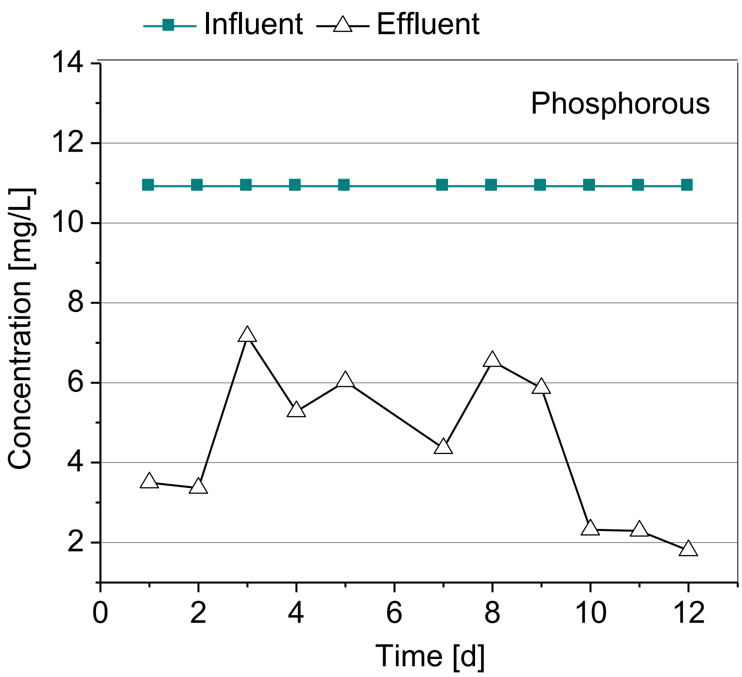

[C]

Figure 5.11 **The trend of ammonium nitrogen, total nitrogen and total phosphorus removal effeciencies with time when the trickling filter is at steady state with respect to organic removal ,[A] to[C]**

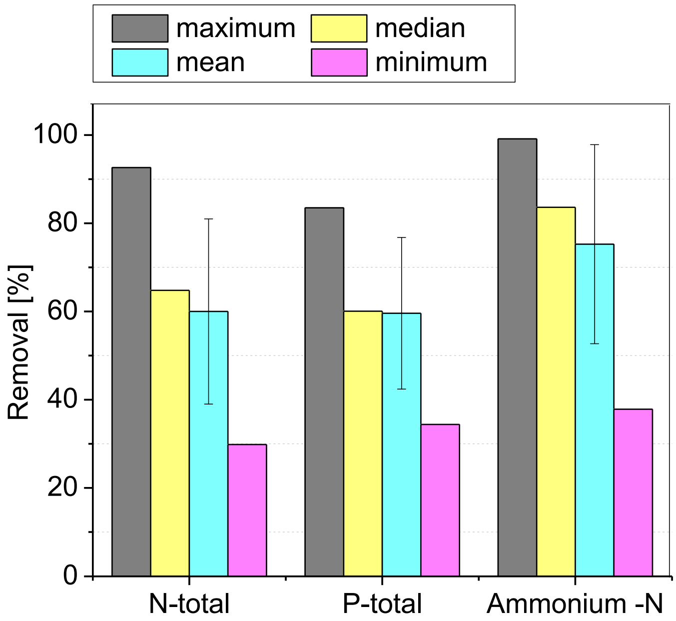

Figure 5.12 Maximum, minimum, and average performance of the trickling filter at steady state

Denitrification

Denitrification was computed as the difference between influent TN and effluent TN, after subtracting biosolids N production. As the cell synthesis g/g theoretical COD is nearly 1 for substrates in brewery wastewater nearly all the total influent COD (0.35) kg will be consumed for cell synthesis. And the theoretical nitrogen consumed will be about (8.75) g for this cell synthesis. During steady state operation, the trickling filter reduced mean influent TN of (41.31) mgL by (92.68) % to produce a mean effluent TN of (3.02) mg/L. Therefore denitrification, as determined from missing N in the mass balance, accounted for (22.28) % of the influent N, assuming 1g biosolids per g of COD and theoretical nitrogen uptake rate of 0.025 g/g of biosolids produced. The result at the different nitrogen removal efficiencies of the trickling filter is summarized in table 5.1.

Table 5.1 Nitrogen removal pathway by the trickling filter

Nitrogen removal [%]	**COD degraded [g]**	**Cell synthesized [g]**	**Nitrogen consumed [g]**		Missed nitrogen
			Theoretical	Actual	
29.85	350	291.55	7.29	3.79	No
60.57	350	291.55	7.29	7.51	No
64.84	350	291.55	7.29	8.04	No
92.68	350	291.55	7.29	11.49	Yes

5.3 Conclusion

The results of the present investigation demonestrated that with respect to total nitrogen removal the efficiency of the trickling is not affected significantly for the given range of hydraulic load i.e. 900 L/d to 1100 L/d, average removal efficiencies ranging from 65.46% to 86.59% could be achieved at influent COD concentration of about 1000 mg/L. High value of organic nitrogen removal by the reactor even in time of organic shock loads could be also achieved by the reactor as can be seen from the result of the operation at different COD loadings. The trickling filter could achieve as high as 92.68 % removal efficiency with regard to removing total nitrogen from the wastewater after the trickling filter reached steady state with respect to COD removal. The flushing away of the nitrifiers at high wastewater flow most probably could not occure in fixed film processes that would otherwise could happen in other conventional treatment systems which could be one reason for un usually high nitrogen removal by the trickling filter.

In removing phosphorus, the trickling filter had performed about 50% in average as the waste water flow rate changes from 900 to 1100 L/d with a slight hydraulic load effect except at hydraulic load of 1100 L/d. With regard to different COD load the bioreactor could achieve removal efficiencies which are in most of the time only slightly varied. The maximum phosphorus removal efficiency was observed at COD load of 1648 mg/L which is about 75 % at a flow rate of 1100 L/d. At the steady state bio film state the trickling filter could achieve average phosphorus removal efficiency of about 80 %. The existence of aerobic and anaerobic biofim zones enhances the removal of phosphorus from the wastewater. Significant reduction of both nitrogen and phosphorus can be achieved together with significant COD reduction as long as the present design and operation conditions are applied.

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6 Start up behavior of the trickling filter during the treatment of industrial brewery wastewater

6.1 Introduction

The factors such as the hydraulic residence time, the reactor's hydrodynamics, the concentration of pollutants in the influent, etc. affect the start up period as well as the performance of fixed-film reactors (Mann, A. et al., 1998). At the start up there might be a need for fixed-film reactors to be inoculated using activated sludge from wastewater treatment plants with suspended growth process (Zhu, S & Chen, S. 2002; Green, M. et al., 2006) or attached bio film from fixed-film reactors. (Zhou, L.et al., 2008) used purchased microorganisms created especially for enhancing municipal wastewater microbiology in fixed-film treatment systems. In a trickling filter bio film can also be established without any special inoculation, only by feeding wastewater to a reactor (Ulug, S. E & Ucuncu, A., 1992; Orantes, J. C. & Gonzalez-Martinez, S., 2003; Mijaylova Nacheva, P. et al., 2008).

According to the report of different researchers start-up of fixed-film reactors may take from 3 to 60 days (Gonzalez-Martinez, S., 2003; Green, M. et al., 2006; He, S.B.et al., 2007; Mijaylova Nacheva, P. et al., 2008; Orantes, J. C.& Gonzalez-Martinez, S., 2003; Y. et al., 2008). Moore et al. (2001) investigated rapid start-up of tested biological aerated filters with respect to suspended solids and chemical oxygen demand removal: in this case the authors reported that the reactors reached steady state bio film system within 3 days, while nitrification was first started on day 20 of the experiment. In contrary to this (Yu, Y. et al., 2008) reported start-up period of 7 weeks for tested biological aerated filters at 20 - 26ºC.

In this chapter of the thesis evaluation of startup performance of a trickling filter filled with locally available gravel as media with respect to removing organics and nutrient from brewery effluents was the aim.

6.2 Materials and methods

The packed media and detailed description of the procedure followed during the packing of the media is given in the previous chapter. The whole research for the present PhD thesis is divided in to two phases. In phase I, investigation of the bioreactor using synthetic brewery wastewater was carried out and in phase II the investigation was conducted using the real industrial wastewater. The investigational data in this chapter takes place in the beginning of phase II operation. The trickling filter was shut down after phase I operation for seven months. The startup behavior of the bio reactor during phase one operation is discussed in detail in chapter three. That implies this chapter is a kind of restart up investigation however this time using industrial brewery wastewater. During this investigation also there was no any external aeration of the bio filter, the same natural phenomenon ensured the aeration of the bio reactor.

6.2.1 Sampling of the local brewery wastewater and analytical techniques

Nearly 200 liters of the wastewater was collected at a time from the brewery factory using cleaned plastic containers of volume 35 liters each. As soon as the wastewater sample reached the laboratory, each container is first mixed thoroughly. Analysis was made by taking a fixed volume of the wastewater from each container and mixing in to a single container. The wastewater sample was analyzed for chemical oxygen demand (COD), biological oxygen demand (BOD_5), nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), ammonium-nitrogen (NH_4-N), total nitrogen (TN), phosphate (PO_4^{3-}), and total phosphorus calorimetrically. The sample preparation and analysis technique of the wastewater parameters is as discussed in detail in the previous chapters. Mixed liquor suspended solids (MLSS) is measured to ensure that there is a sufficient quantity of active biomass available to consume the applied quantity of organic pollutant at any time.

MLSS is the concentration of suspended solids, in the secondary clarifier tank during the trickling filter process. MLSS consists mostly of microorganisms and non-biodegradable suspended matter. This is known as the food to mass ratio, more commonly notated as the F/M ratio. By maintaining this ratio at the appropriate level the biomass will consume high percentages of the organic matter in the wastewater. Therefore MLSS was measured by filtering 100 mL samples of the liquid phase and drying at 105°C for 24 hours (data not presented here).

6.2.2 Pilot scale trickling filter and operation of the trickling filter

The detailed description of the pilot scale trickling filter is given in the previous chapters.The only modification made during the operation of the trickling filter with the local brewery wastewater is the inclusion of pre settling basin as this time the wastewater might contain suspended solids. The modified flow scheme of the bioreactor is given in Fig. 6.1.

Representative wastewater sample was collected from the equalizing chamber of the brewery with cleaned plastic containers each 30 L volume. About 200 L of the wastewater was collected and transported to the laboratory every day in which the pilot scale trickling filter was installed and the research takes place. To evaluate the acclimation behavior of the trickling filter for COD, total nitrogen and total phosphorus about 100 L of the brewery effluent with a pH ranging from 7-8 and temperature 19-21°C was fed to the trickling filter every day. The influent concentration of COD, total nitrogen, and total phosphorus was increased gradually from 269-3836,1.6-125 and 1.05-23.8 respectively during this investigation, here only the initial concentration of COD was monitored and the resulting change in initial concentration for total nitrogen and total phosphorus was due to the fluctuation of the wastewater composition.

Then after concentration reduction was monitored to evaluate the growth of the microbial population. Microscopic visualization of the microbial population was also carried out during this time. The achivement of the acclimation period was realized when the analysis for three consecutive days on the final effluent were approximately the same during the operation of the trickling filter at the highest COD load. The trickling filter was operated at ambient wastewater temperature (19 – 21 ºC). The fed wastewater was intermittently sprayed over the trickling filter every day with 1.50 h per interval.

In order to avoid the development of passage along a trickling filter medium that allows the applied wastewater to pass only in one direction influent wastewater should be distributed over a medium as uniformly as possible (Wik, T., 2003). For uniform distribution of the wastewater the end portion of the tube through which the feed wastewater comes was perforated and coiled on top of the trickling filter after placing the sieve plate under it. Removal efficiencies of the trickling filter was calculated based on the reduction of concentration between the influent and effluent streams as shown in Eq.(1).

$$\text{Removal (\%)} = (C_{in} - C_{out}) / C_{in} \times 100\% \qquad (1)$$

where C_{in} was influent concentration [mg /L] and C_{out} effluent concentration [mg /L].

6.3 Results and discussion

6.3.1 Wastewater composition

The composition of the brewery wastewater characterized during this phase of the trickling filter operation is given in Table 6.1. The analysis characterized the brewery wastewater from the Rostock brewery as having a BOD_5, COD, NH_4−N, N−total, PO_4^{-3}−P and P− total concentration range of 1.1−8.66 g/L, 1.345−11.026 g/L, 0.034−0.23 g/L, 0.004−0.038 g/L, 0.0098−0.073 g/L, respectively. Values of all compounds that exist in ionic form stated throughout are soluble values only and were determined from micro filtered samples using syringe filter that has a pore size of 0.45 µm. Nitrate and nitrite concentration in the wastewater was near to zero in most of the analysis. The BOD_5 to COD ratio during the different sampling dates ranges from 0.69 to 0.87.

Table 6.1 Physicochemical properties of the wastewater on different days of the week

COD [mg/L]	BOD_5 [mg/L]	NH_4 [mg/L]	NO_3N [mg/L]	NO_2N [mg/L]	TN [mg/L]	PO_4-p [mg/L]	TP [mg/L]	T[°C]	pH
1650	1144	3.63	0.654	0.586	31.0	8.33	11.64	17	8
3949	3444	8.21	3.23	0.173	55.8	5.63	14.3	31	9
11023	8664	13.8	11.1	0.588	229	38.2	73.2	26	12
1599	1215	3.25	16.9	0.590	44.1	5.88	11.64	26	8
1970	1548	2.77	2.32	0.433	34.1	3.6	9.84	32	9
1345	1100	3.63	0.654	0.586	31.6	8.33	11.64	17	8
9217	4245	2.17	19.6	0.221	128	26.5	38	25	11
4225	3500	4.44	3.19	0.096	58.7	4.55	14.5	34	10
2258	1921	4.83	3.22	0.063	33.9	5.88	9.85	26	8

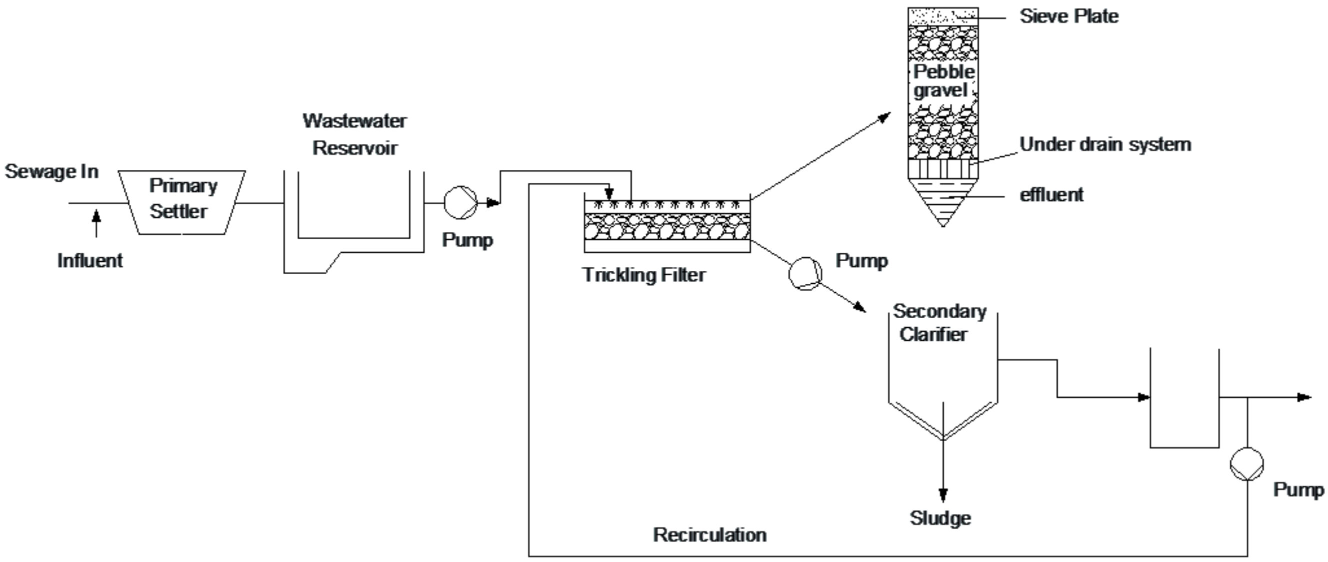


Figure 6.1 The modified flow scheme of the trickling filter

6.3.2 Trickling filter operation and start up period

During the present operation of the trickling filter the influent COD concentration of the wastewater increased from 269 mg/L to 3836 mg/L mean while total nitrogen and total phosphorus influent concentration changed from 1.6 mg/L to 125 mg/L and 1.05 mg/L to 23.8 mg/L respectively. The trickling filter was kept at each concentration of the feed water for 24 hours while feeding intermittently the influent and recirculating simultaneously. The removal efficiency ranges were 71.22 % to 87.42%, 25.54% to 68.53% and 17.12% to 87.5% for COD, total nitrogen and total phosphorus respectively as a result.

The trickling filter stabilized with respect to the removal of COD within about two weeks with maximum COD removal efficiency of 87.42%. At this steady state bio film cover the efficiencies of removal for TN and TP were, 68.53% and 87.5% respectively. The trend curve for reduction of COD, TN and TP during the acclimation time of the bioreactor is shown in Fig.6.2. And the raw data for the figures are given in Annexes A12 to A14.

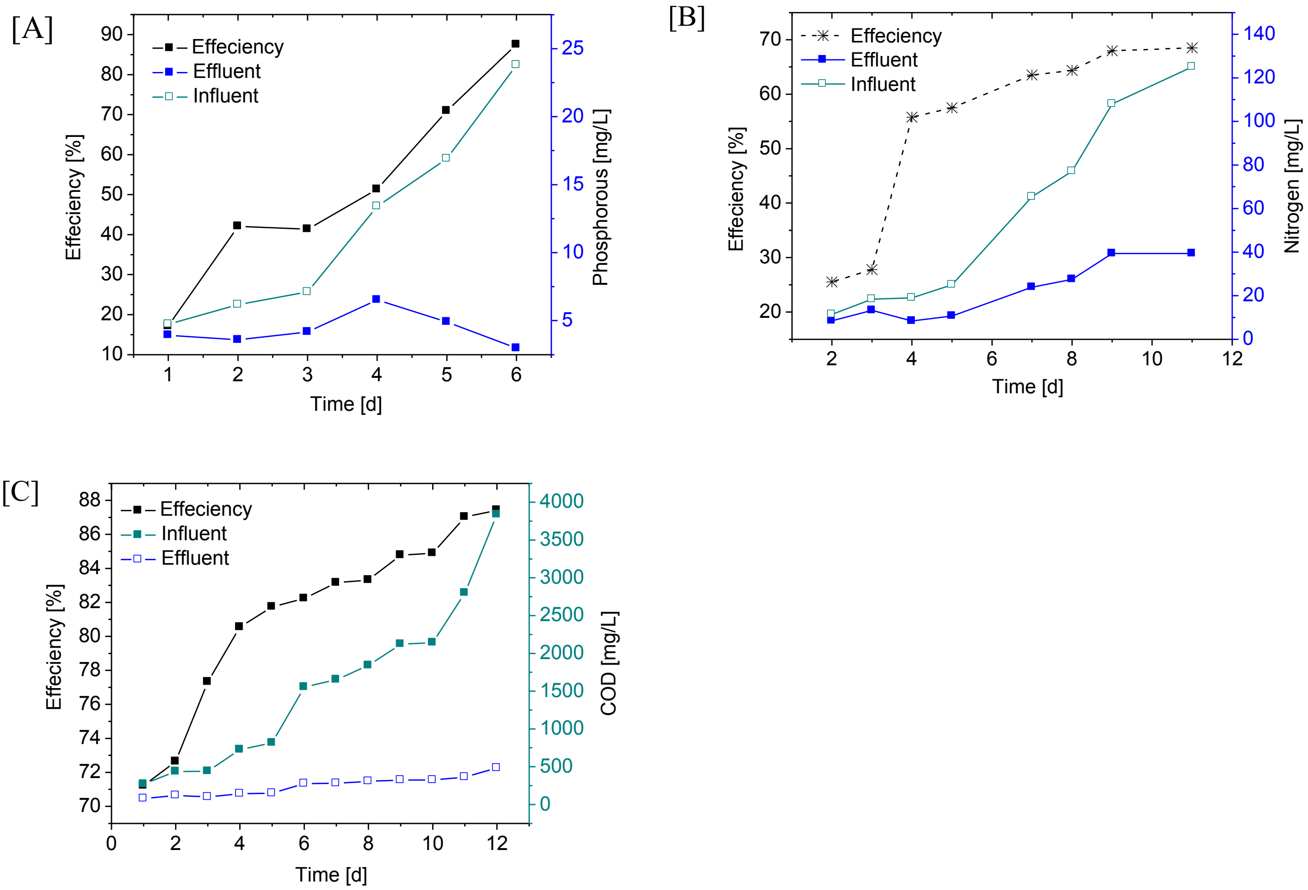


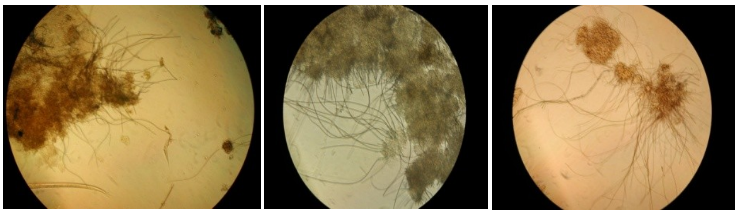
Figure 6.2 **Concentration reductions by the trickling filter during the acclimation phase ,[A] to [C]**

As per other researchers start-up of fixed-film reactors may take from 3 to 60 days (Gonzalez-Martinez, S., 2003; Green, M. et al., 2006; He, S.B. et al., 2007; Mijaylova Nacheva, P. et al., 2008, Orantes, J. C.& Yu, Y. et al., 2008). In this investigation the start up of the bioreactor achieved without the need for special inoculation and the start up period was only about 15 days. The success of rapid bio film development on the trickling filter medium without special inoculation and at ambient temperature indicates that the substrates in the wastewater itself with the optimum temperature and pH of the wastewater is suitable as standard nourishment for the indigenous microbial populations to originate from the wastewater and multiply on the trickling filter medium. The very high BOD to COD ratio of brewery wastewater as also proved during the wastewater analysis of the local brewery and the standard temperature and pH condition that was maintained during this period plays role for the rapid start up period. The other condition that enhanced the rapid start up could be the type and geometry of the packing media used in the trickling filter. Fig. 6.3 depicts picture of some of the microbial populations of the trickling filter viewed under light microscope during the time of its full operation.

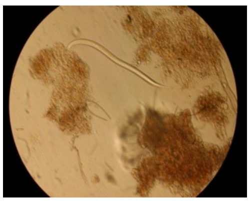
6.3.3 Microbial populations in the trickling filter

The organisms that can be found in a trickling filter do not differ much from the ones found in activated sludge. No differences have been found between the bacterial flora of activated sludge and that of trickling filters (Lin, 1984). In activated sludge most are attached to suspended flocs, whereas in a trickling filter they attached to the filter bed. Apart from heterotrophic and nitrifying bacteria, the following organisms take part in the purification process (see Mudrack and Kunst (1986), Tri (1975), Fair et al., (1968)): Zooflagellates (*Mastigophora*), especially in highly loaded systems, Amoebae, different species appear in differently loaded systems, Ciliates, they are very common and they graze on bacteria. There are attached species, like representatives of the genus *Vorticella*, and free swimming species, belonging to genera like *Aspidisca*, *Paramecium* and *Euplotes*. Nematoda Diatoms, usually are present in slightly loaded systems.

a) A zooflagellate, *Bodo sp.*



b) Filamentous bacteria



c) A nematode

Figure 6.3 Some organisms, encountered in the trickling filter bio film

6.4 Summary and Conclusions

Huge amount of process water that is high in COD and nutrients is the major environmental challenges of breweries. The commonly available treatment technologies for wastewater discharges from brewery company have high installation and operation cost, have complex operation, high electricity demand and not capable of removing organics and nutrients simultaneously. In the present investigation the capacity of a mineral filled trickling filter which is not capital intensive and needs lower operational and maintenance cost is evaluated for start up behavior.

The start up of the bioreactor can be achieved at ambient temperature in two weeks of operation. At the start up efficiency of COD removal was higher than 87 %, while removal efficiency of nitrogen and phosphorous was higher than 68 % and 87 % respectively. It was not required to raise the pH of the wastewater for the slow growing nitrifiers as more than 50 % of nitrogen removal was achieved starting from the 4th day of operation. A locally available gravel medium can be used in the trickling filters without a problem of clogging in such elongated trickling filter tanks also. Therefore the proposed biological treatment process can be ready for full operation with a minimum operation cost and relatively rapidly.

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7 The efficiency of the trickling filter at different hydraulic and organic loading rate during the treatment of industrial brewery wastewater

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Abstract

Despite sound technological improvements, huge water consumption and discharge remain one of the major environmental challenges in the brewing industry. The aim of this study is therefore to treat industrial brewery wastewater using a trickling filter. A trickling filter is superior over the other conventional wastewater treatment processes in terms of cost and environmental friendliness therefore it is recommendable for low income countries.

Pilot scale plant experiments were made to evaluate the trickling filter aerobic and anaerobic bio film systems for removal of organics and nutrients from brewery wastewater at different hydraulic loading rate. The trickling filter had an average efficiency of 86.53 %, 95.25 %, 69.93 % and 41.03 % for biochemical oxygen demand (COD), biological oxygen demand (BOD_5), total nitrogen (TN) and total phosphorus (TP) respectively as the flow rates changed from 900 L /d to 1100 L /d and at influent COD concentration of 600mg/L. Mass loading rate and mass removal rate are highly correlated as can be proved from the linear regresson model, correlation coeffecients (98 %, 99 %, 85 % and 65 % for COD, BOD_5, nitrogen and phosphorus). The result of this study also suggests that in addition to significant values for nutrient removal efficiencies, effluent value in the range of 50 mg/L − 120 mg/L COD can be achieved using the trickling filter during the treatment of industrial brewery wastewater at the design hydraulic load and organic load of 8.36 $m^3/m^2/d$ and 0.75 kg $COD/m^3/d$ respectively. Organic matter and nutrient content of brewery wastewater can sufficiently be reduced using a single stage rock filled trickling filter.

Key words: Bio films; Biomass; Aeration; Removal; Recycling; Sludge

7.1 Introduction

Industrial discharge into rivers is one cause of irreversible degradation occurring in surface water systems (Rajaram and Ashutosh, 2008). Due to their role in carrying off industrial wastewater, rivers are among the most vulnerable water bodies to pollution. There have been significant impairments to rivers from pollutants, rendering the water unsuitable for beneficial purposes, such as domestic use, irrigating agricultural lands, recreation, drinking, wildlife propagation, and food processing purposes in industries; all of these uses are on the rise, particularly in developing urban areas. With increasing scarcity of a treated public water supply, fresh river water has become an alternative source for these purposes (van der Bruggen and Braeken, 2006). Of such industries breweries may be the largest consumer of water and the largest source of organic effluent that must be treated by the municipal treatment plant.

The effluents discharged from breweries are found to have high organic and acidic content, which increases the BOD, COD and high organic load in the waste water contributive to dissolved carbohydrates, alcohols, suspended solids, yeast etc, which pollutes the water bodies considerably (Chaitanya Kumar et al., 2011).

The performance of the trickling filter to remove organics and nutrients from synthetic brewery waste water has been reported in chapter three and chapter five in detail. In the present chapter the performance evaluation is using industrial brewery wastewater.

7.2 Materials and methods

7.2.1 Trickling filter operation

Representative wastewater sample was collected from the equalizing chamber of the brewery with cleaned plastic containers each 30 L volume. About 200 L of the wastewater was collected and transported to the laboratory at one time in which the pilot scale trickling filter was installed and the research takes place. Intermittent mode of operation with recirculation was the type of operation throughout the experiments. During all operation of the trickling filter temperature was varying around 20°C. The fed wastewater was intermittently sprayed over the trickling filter (Tf) every day with 1.50 h per interval. Removal efficiencies of the trickling filter was calculated based on the reduction of COD concentration between the influent and effluent streams as shown in Eq.(1)

$$\text{COD removal (\%)} = (C_{in} - C_{out})/\ C_{in} \times 100\% \qquad (1)$$

Where as hydraulic loading rate, mass loading rate and mass removal rate for the trickling filter are calculated using Equations (2) – (4):

HLR ($L/m^2/d$) was:

$$\text{HLR} = \frac{Q}{A} \qquad (2)$$

where Q is the wastewater flow including the recirculation flow (L/d) and A is cross-sectional surface area of the trickling filter.

Mass loading rate (MLR) and mass removal rate (R_{mass}) of wastewater pollutants were calculated in g m^{-2} d^{-1}.

$$\text{MLR} = \frac{C_{in}\ \text{X HLR}}{1000} \qquad (3)$$

$$R_{mass} = \frac{(C_{in} - C_{out})\text{X HLR}}{1000} \qquad (4)$$

where C_{in} was influent concentration (mg /L), and C_{out} effluent concentration (mg/L).

7.2.2 Instrumentation and analytical methods

The wastewater samples were then analyzed for chemical oxygen demand (COD), biochemical oxygen demand (BOD_5), nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), ammonium-nitrogen (NH_4-N), total nitrogen (TN), phosphate (PO_4^{3-}), and total phosphorus colorimetrically. Prior to analysis the soluble fraction of the wastewater samples were obtained by filtering with a syringe filter (25mm diameter (W/0.45µm cellulose) for analysis of nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), ammonium-nitrogen (NH_4-N), and phosphate (PO_4^{3-}). An unfiltered wastewater sample was used for analysis of COD, total nitrogen, and total phosphorus.

Biomass concentrations were determined by weighing dried (24 h, 105 °C) 100 mL samples of the liquid phase. A Spectrophotometer (Hach Lange Xion 500 LPG385) was employed for the measurements of COD and nutrients. A thermostatically controlled incubator with standard /glass door was employed for the measurement of BOD_5. Microprocessor controlled standard-pH-ion-meter pMX 3000 / pH was used to measure the pH and temperature of the influent and effluent. The influent and effluent samples collected and kept in a refrigerator were analyzed for the selected parameters using the Dr. Lange cuvette test system as a reagent. During all experiments, COD concentration, ammonia nitrogen, phosphate phosphorous, total nitrogen, pH and temperature measurements were made three times per week unless otherwise indicated.

7.3 Result and discussion

To investigate the response of the trickling filter to change in wastewater flow four different values, namely 1100, 1050, 950 and 900 [L/day] (at a hydraulic load of 8.76 $m^3/m^2/d$, 8.40 $m^3/m^2/d$, 7.56 $m^3/m^2/d$, 7.2 $m^3/m^2/d$) including recirculation flow were chosen. Influent concentration of COD, BOD_5, TN and TP were all kept similar in all the investigations and temperature and pH varied around 20°C and 7 respectively. The ranges of efficiencies (%) for COD was (84 - 92.2), (69 - 87.7) (84.8 - 95.5) and (64.7 - 90.9) when the hydraulic loading rates varied as 7.17, 7.56, 8.36 and 8.78, [$m^3/m^2/d$] respectively. Where as the ranges of efficiencies (%) for total nitrogen and total phosphorus respectively were (11.49 - 63.87, 10.53 - 71.28, 41.63 - 83.27 and 27.58 - 75.59) and (34.05 - 40.06, 19 - 30.37, 30.32 - 56.48, 15.22 – 43.40). Fig.s 7.1 to 7.4 illustrate the trend of change of concentration of the treated wastewater during the operation of the trickling filter at the four different hydraulic loading rates. Where as the raw data for these figures are given in Annex A15 to A16. The average efficiencies (mean ± SD) for COD, BOD_5, TN and TP as the hydraulic loading changed from 7.17 $m^3/m^2/d$ to 8.78, $m^3/m^2/d$ is illustrated in Fig.7.5. And the raw data table is given in Annex A17.

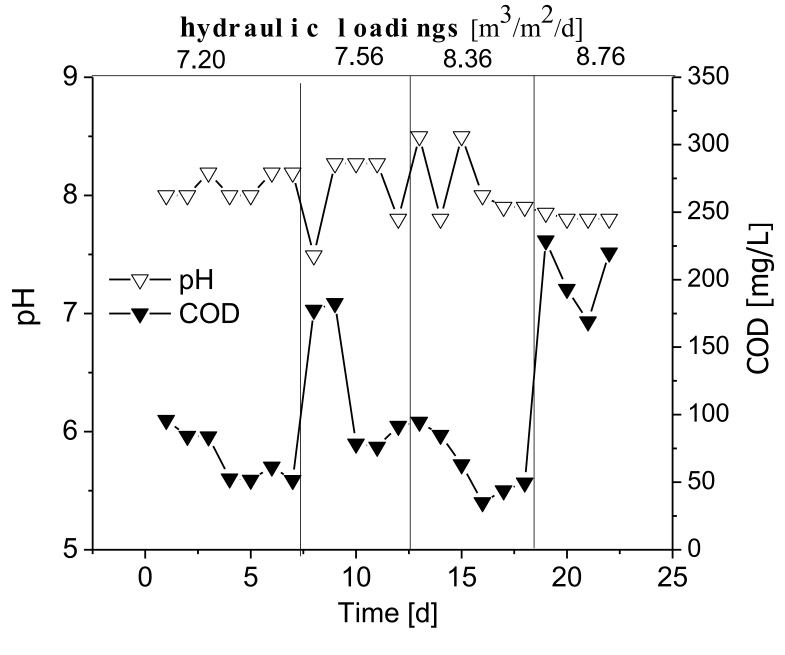


Figure 7.1 Effluent properties of the trickling filter as a function of hydraulic loadings (Influent COD = 621±60.25 mg/L)

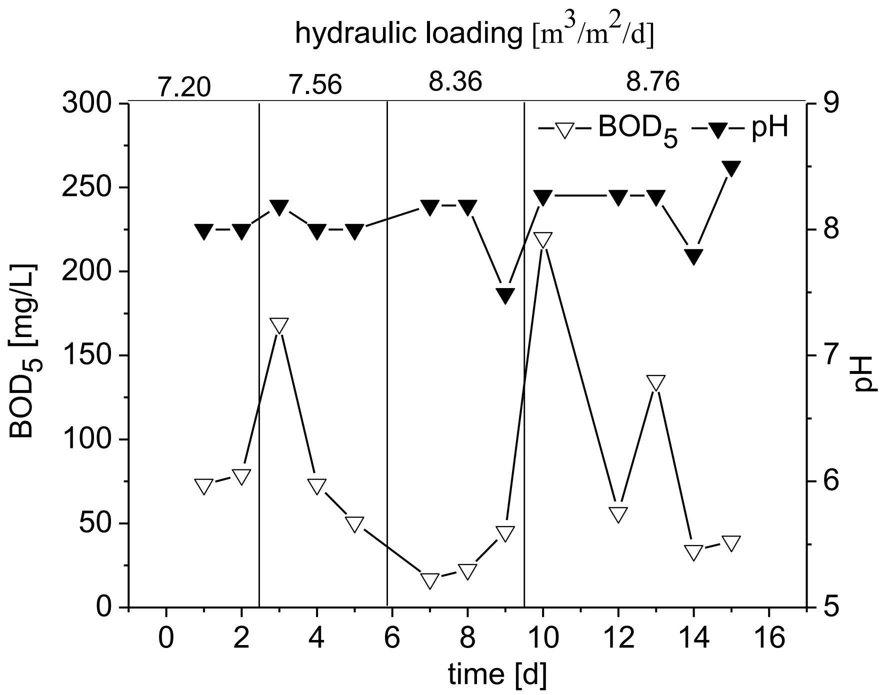


Figure 7.2 Effluent properties of the trickling filter as a function of hydraulic loadings (Influent BOD_5 = 577±52 mg/L)

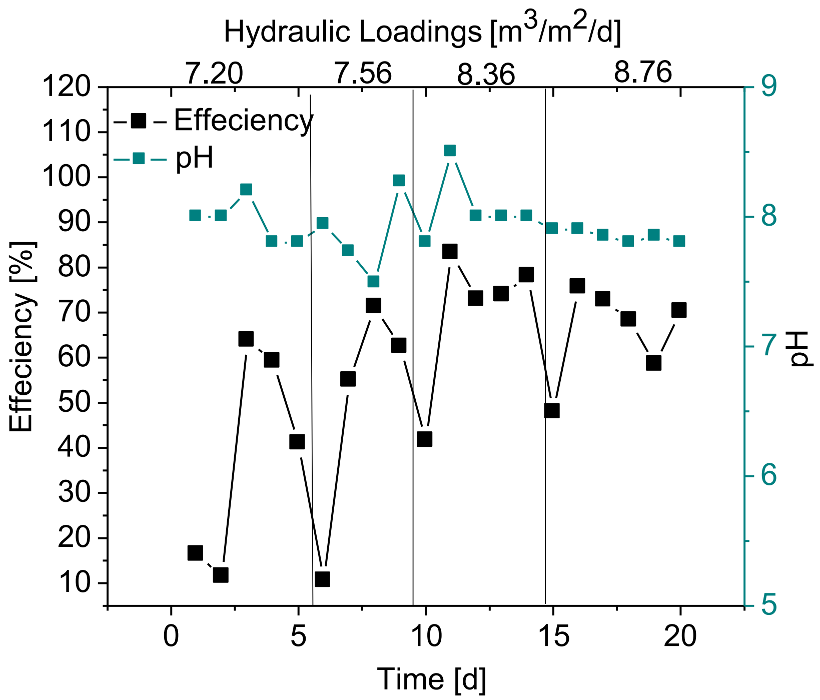


Figure 7.3 TN removals at the different hydraulic loadings in the trickling filter

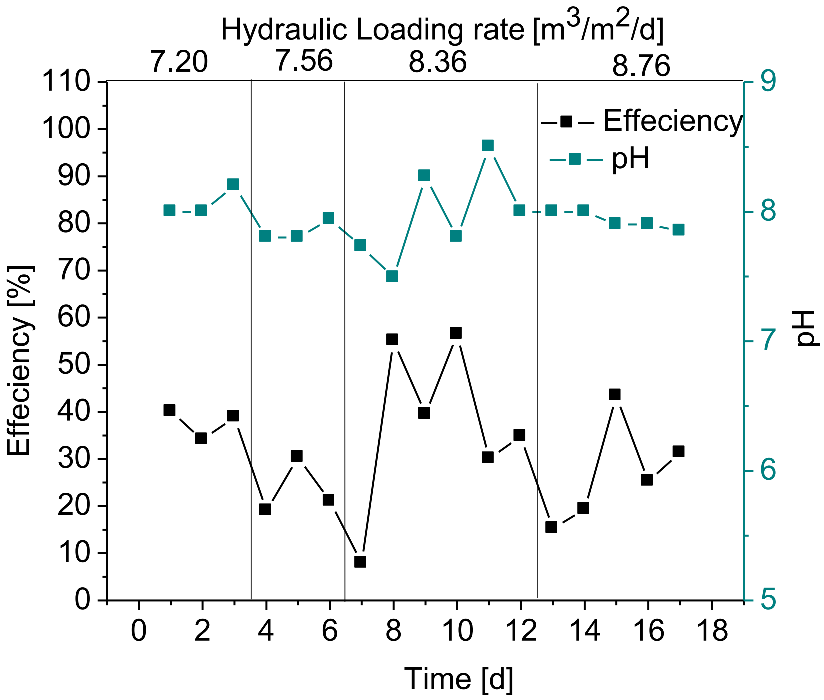


Figure 7.4 **TP removals at the different hydraulic loadings in the trickling filter**

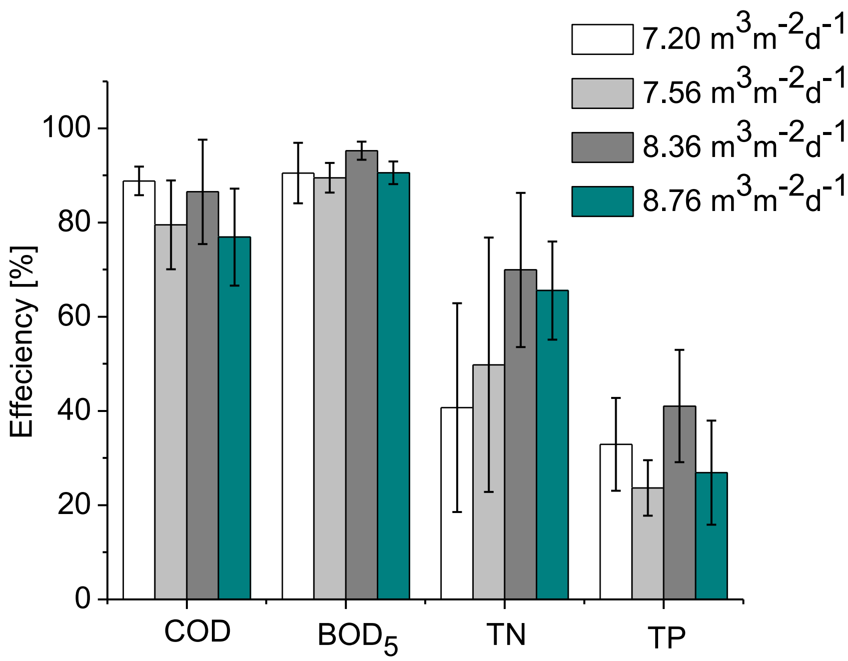


Figure 7.5 **Average efficiencies of the trickling filter at different hydraulic loading rate**

Except for total nitrogen there is a slight decrease generally in efficiency of the trickling filter with hydraulic loadings which is attributed to the reduced residence time of the wastewater as the flow increased, a lesser contact time of the wastewater parameters with the the bio film will happen. The other reason is the higher scour for media surfaces with increase in hydraulic loading rate. In this investigation except slightly lower efficiency for COD impresively the trickling filter achieved the maximum effeciencies for all COD, BOD_5, nitrogen and phosporous at hydraulic loading rate of 8.36 $m^3/m^2/d$. The exceptional high effeciency at high hydraulic loading rate is attributed to the increased microbial development and increased activity of microbial population due to the concomitant increase in organic loading with increase in hydraulic loadings. In addition the bio film thickness control at the higher hydraulic load will reduce the internal diffusion limitation thereby enhancing substrate transfer (Westerman et al, 2000; V. Lazarova, J. Manem, 1994). Enhanced aeration at higher hydraulic load can also be the other reason. Nevertheless as attached growth systems are characterized by having high sludge concentration, unlike the other biological treatment methods high removal rates at relatively small hydraulic retention times is exhibited by the systems. The average efficiencies of the trickling filter for $BOD_{5,}$ COD, TN and TP at 8.36 $m^3/m^2/d$ were 95.25 %, 86.53 %, 69.93 % and 41.03 % respectively. And average efficiency of the trickling filter for $BOD_{5,}$ COD, TN and TP at the maximum value of hydraulic loading rate i.e. at 8.76 $m^3/m^2/d$ was 90.58 %, 76.90 %, 65.55 % and 26.89 % respectively. This leads to a conclusion that trickling filter is an ideal treatment technology in time of wastewater flow variation which is typically encountered in brewery industry.

Unlike removal efficiency removal loading rate of the trickling filter increased with hydraulic loading rate for all the parameters (Figures 7.6 and 7.7). Except the very small decrease of mass removal rate for total phosphorus and BOD_5 as the hydraulic loading rate (HLR) changes from 1.99 to 2.39 [$m^3/m^2/d$] , the mass removal rate of the trickling filter increased as the hydraulic loading rate increases from 0.80 to 2.39 [$m^3/m^2/d$] for all parameters i.e BOD_5 from 406.74 $g/m^2/d$ to 1090.76 $g/m^2/d$; total nitrogen from 3.58 g /m^2/d to 26.35 $g/m^2/d$; COD from 437.21 $g/m^2/d$ to 1248.73 $g/m^2/d$; and total phosphorus from 0.77 $g/m^2/d$ to 2.29 $g/m^2/d$ (Annex A18 to A19). Depending on these observations the optimum hydraulic loading rate for simultaneous COD, $BOD_{5,}$ nitrogen and phosphorus removal efficiency and removal loading rate for all the parameters is 8.36 $m^3/m^2/d$.

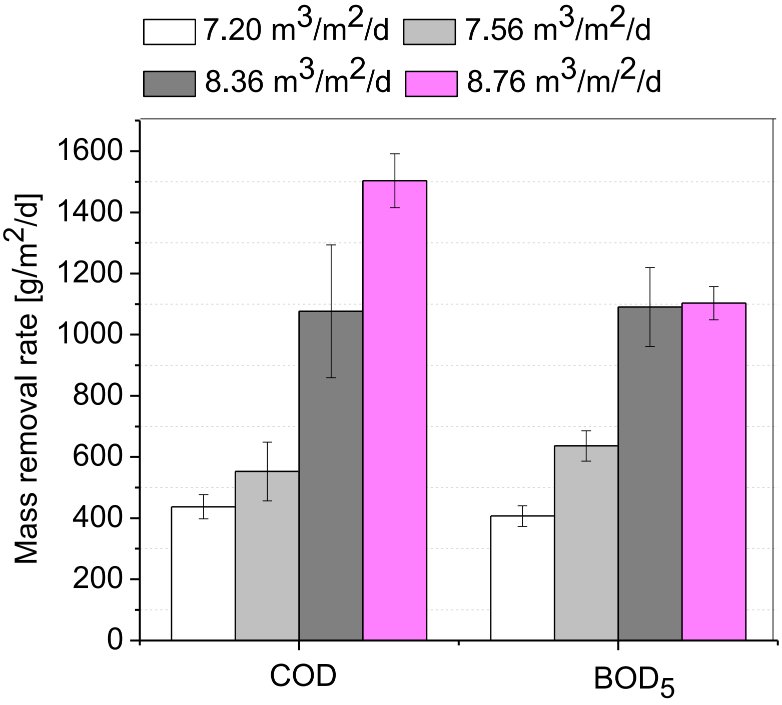


Figure 7.6 Mass removal rate of the trickling filter at diffrent hydraulic loading rate

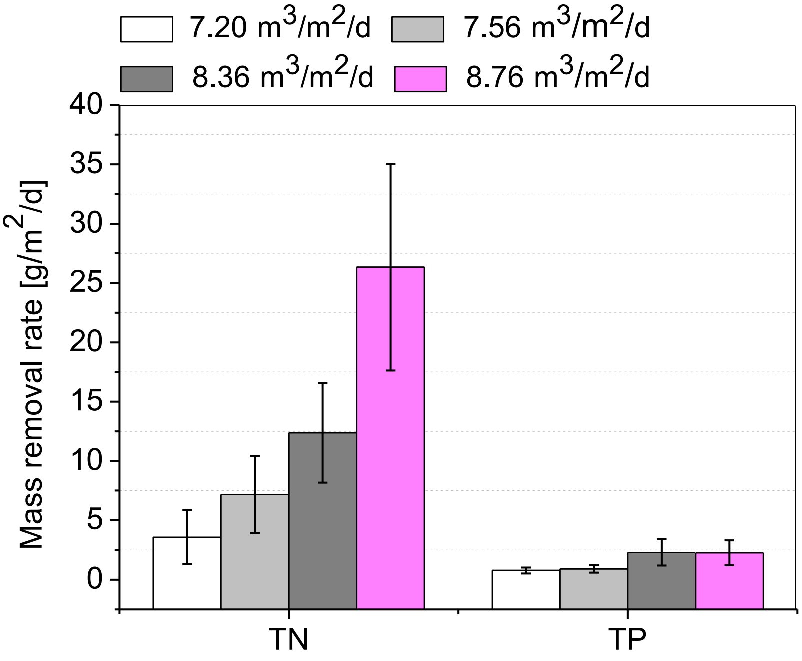
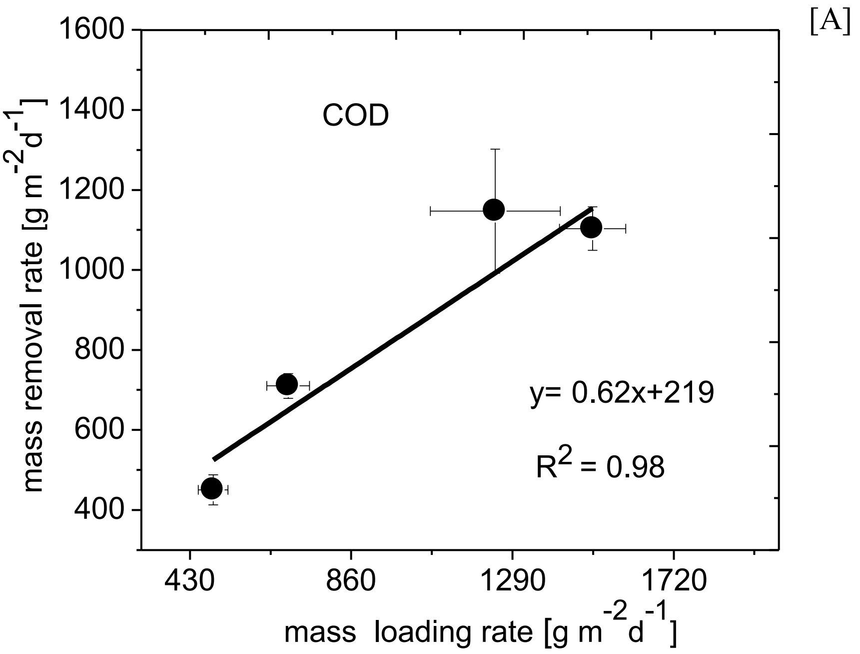
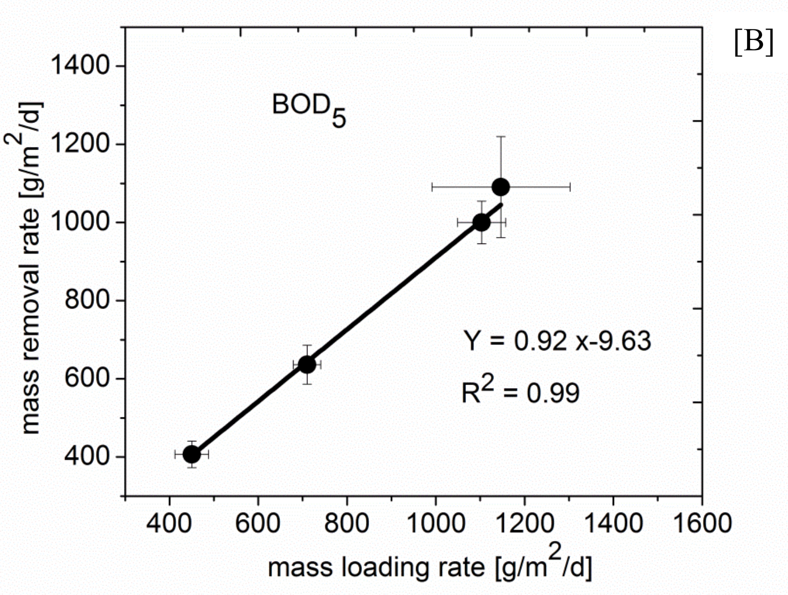
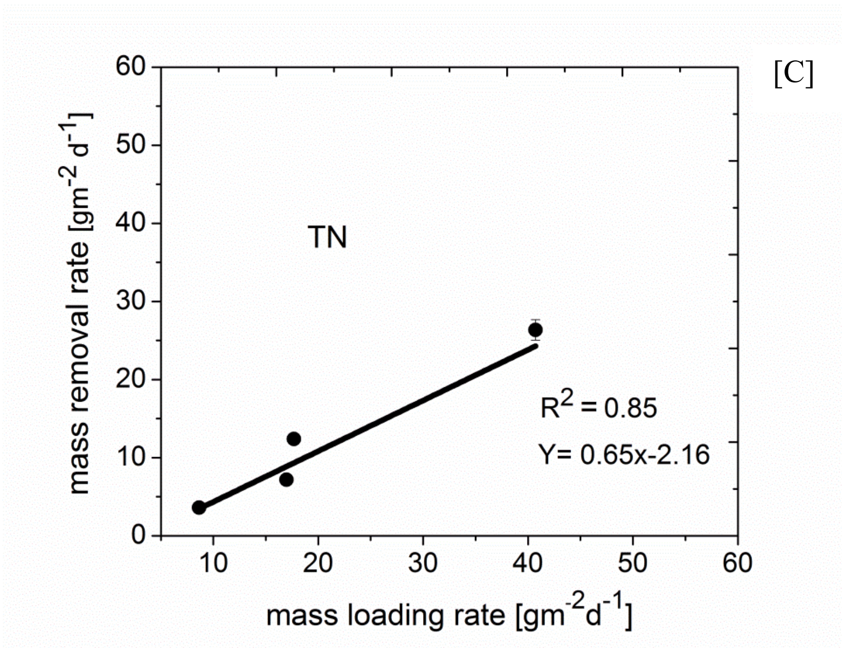


Figure 7.7 Mass removal rate of the trickling filter at diffrent hydraulic loading rate

Linear regression model relating mass loading rate and mass removal rate is illustrated in Fig.7.8. Except for total phosphorus there is good correlation between the mass loading rate and mass removal rate. The R square value for COD and BOD_5 removal is near to one which suggests it is highly probable that the removal capacity of the trickling filter increase with mass loading rate even when the hydraulic loading rate increase beyond the investigated range.







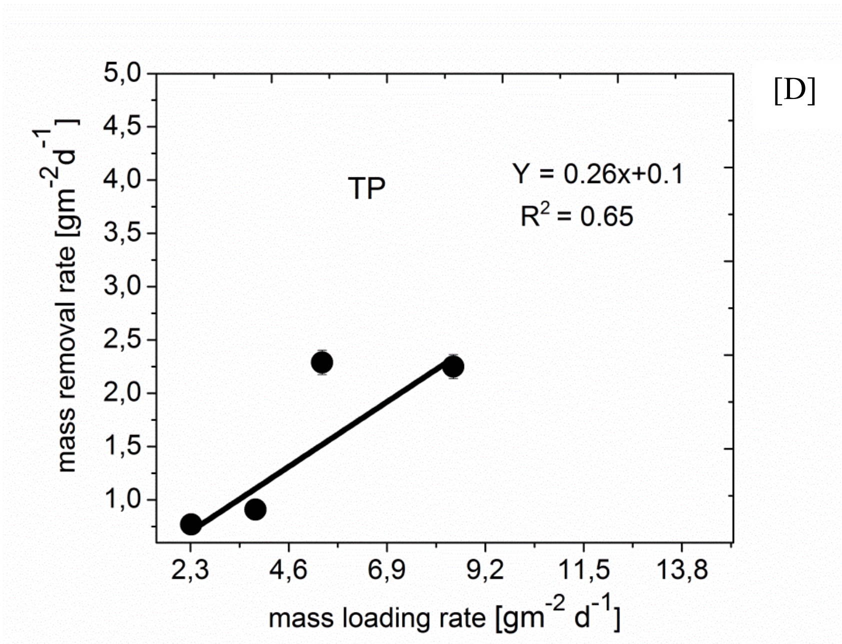
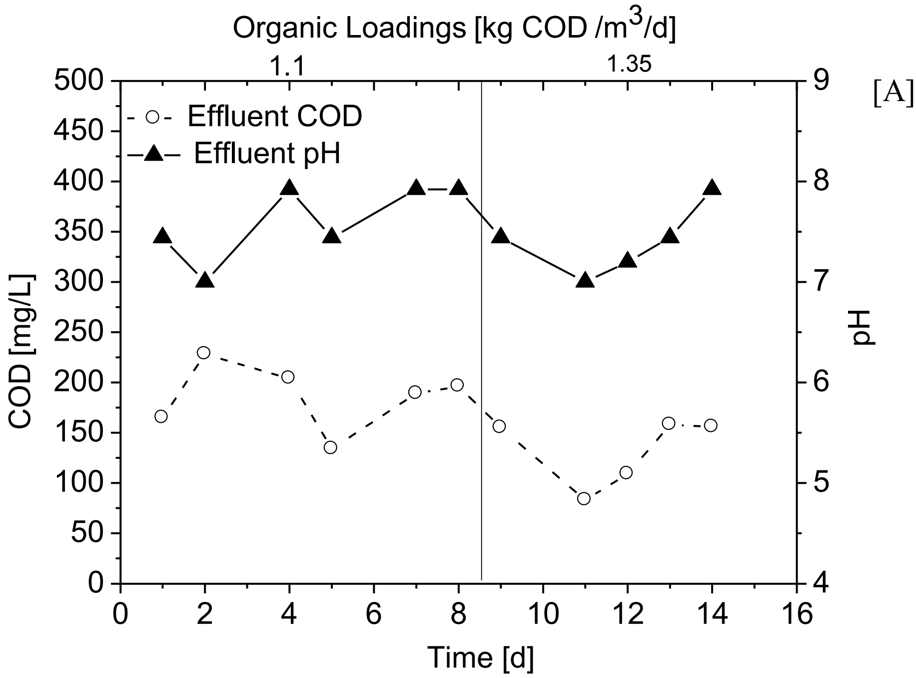
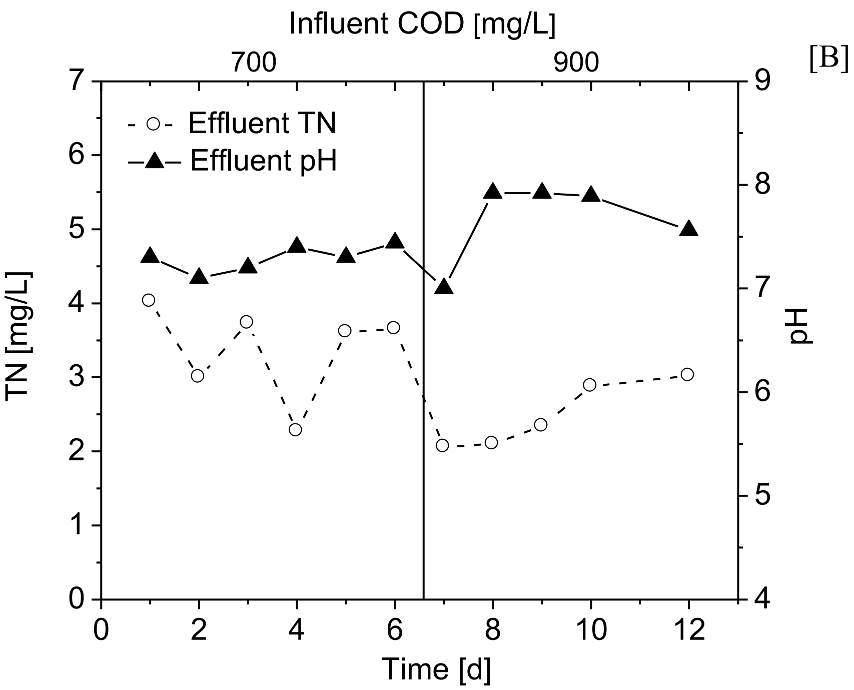


Figure 7.8 Linear regression analysis comparing observed mass removal rates and mass loading rates (both in g/m^2/d), [A] to [D]

The development of highly concentrated bacterial biomass by the trickling filter medium also confirmed from the operation of the trickling filter at two different COD loadings varying at its design COD loadings (Figure 7.9). During this operation the trickling filter achived higher removal effeciencies in the removal of all COD, BOD_5, nitrogen and phosphorus at the higher COD loadings (Figure 7.10). Further the raw plotted data is given in Annex A20 and A21


Organic Loadings [kg COD /m3/d]
1.1
1.35
[A]
Effluent COD
Effluent pH
COD [mg/L]
pH
Time [d]


Influent COD [mg/L]
700
900
[B]
Effluent TN
Effluent pH
TN [mg/L]
pH
Time [d]

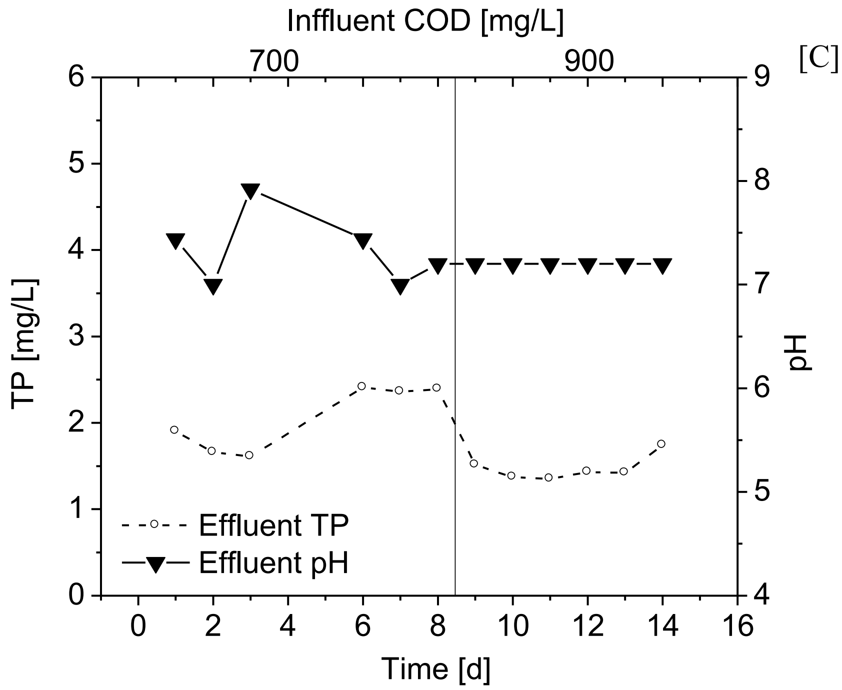


Figure 7.9 **Effluent property of the trickling filter. Operating in a continuous mode at constant Q_{feed} of 300 L/d and two different design COD load, [A] to [C].**

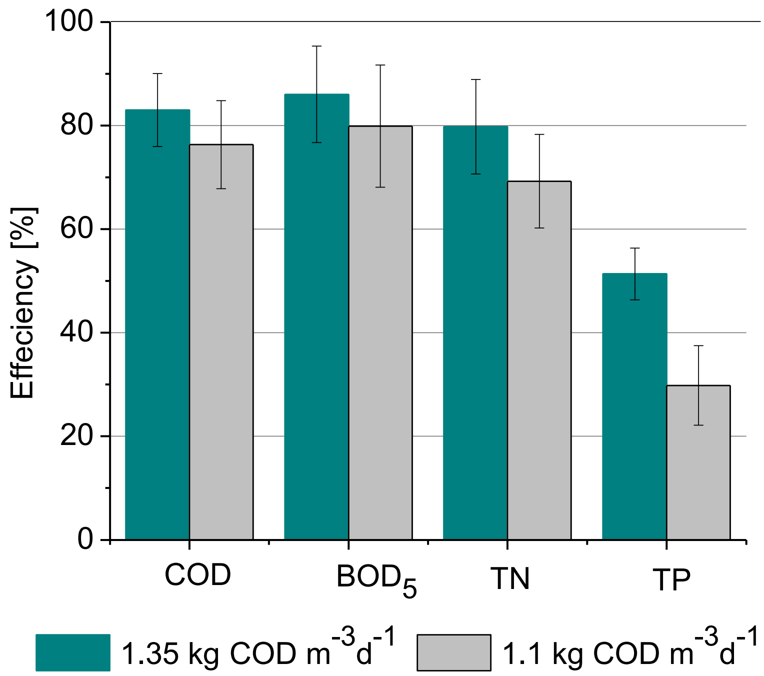


Figure 7.10 **Average efficiencies of the trickling filter for organic and nutrient removal at two different organic loading rates.**

7.4 Summary and Conclusion

Start up of the pilot scale trickling filter (1.8 m in height and 0.4 m in diameter) using the river stone substrate medium at ambient wastewater temperature (19 ºC – 21 ºC) was investigated over a period of 30 days. A rapid start up period without a need for special inoculation was achieved by the trickling filter.The acclimation period was only about two weeks for all COD, total nitrogen (TN) and total phosphorus (TP).

It is found from the investigation of the effect of hydraulic loading rate that there was only very slight decrease in efficiency with increase in hydraulic loading rate with respect to the reduction of all wastewater parameters. Further more the trickling filter achieved maximum removal efficiency at near the maximum hydraulic loading rate for all parameters.This can lead to a conclusion that high hydraulic loading rate has a possetive effect too on the efficiency of the trickling filter. The correlation coeffecients from the linear regression models suggested mostly the mass removal rate increase with mass loading rate except in the removal of phosphorous.That confirms the existence of high biomass concentration in the trickling filter. The near to one correlation coefficients for COD, BOD_5 and TN for linear regression curve means that it is highly probable that the mass removal rate increase with mass loading rate even when the hydraulic loading rate increased beyond what is investigated in this study. The development of highly concentrated bacterial biomass by the trickling filter medium also confirmed from the operation of the trickling filter at two different COD loadings varying at its design COD loadings. The trickling filter achived higher removal effeciencies of all COD, BOD_5, nitrogen and phosphorus at the higher COD loadings.

To conclude from the present investigation, a single stage trickling filter without a need to integrate it with other treatment technologies and without external aeration can be utilized to reduce the organic and nutrient load of a brewery wastewater in a cost-effective and environmentally friendly manner .

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8 Economic evaluation of the trickling filter and other conventional sewage treatment processes

8.1 Introduction

In developing countries such as Ethiopia, the rate of sewage service utilization is generally still very low due to high costs of WWTP (wastewater treatment plant). WWTP costs are in general subdivided in to investment and operating costs. The latter may be fixed (normal operation and maintenance, fixed power) or variable (power and chemical consumption, sludge treatment and disposal and effluent taxes). The cost of a sewage treatment process varies significantly depending on the time frame and location. Moreover, the configuration of any similar type of treatment process may vary according to the size of the local community or climatic conditions of the area, which in turn affects cost. These factors considerably affect the task of standardizing the cost of any process.Consequently; it is difficult to define a given process with marginal cost, which is important for developing countries.

However, it may be possible to understand the general trend of such costs as estimated by several researchers, especially for developing countries. Table annex C1 summarizes the capital, operation and maintenance (O&M) costs, and land requirements for up-flow anaerobic sludge blanket (UASB), waste stabilization pond (WSP), and activated sludge process (ASP) treatment systems. These costs and land requirements are often expressed by the following equation (Balmer and Mattsson, 1994; JSWA, 1999; JSWA, 2001; Li, 1987; Li et al., 1990; Tsagarakis et al., 2003):

$$\text{Cost per unit volume} = a \times (\text{size})^{b},$$

where a, b are the constants. The value of constant avaries when substituting "size'' with a population equivalent or treatment volume, and depending on the type of currency to be expressed. The variation of cost per unit volume by size for a process is indicated by the constant b. Table annex C1 describes the scale merit of ASP. For example, the Japan Sewage Works Association reported a cost model of ASP in which the values of the constant b are -0.28 for capital and -0.19 for O&M cost (JSWA, 2001). Balmer and Mattsson (1994) estimated the value of constant b as -0.3 for O&M cost according to 20 STPs (sewage treatment plants) treated by an ASP system in Sweden.

8.2 Economic evaluation of the trickling filter

Achieving an effluent standard using an industrial wastewater treatment processes that are cost effective are generally preferred by any country, especially developing countries. The main expenses are capital cost, operation and maintenance (O&M) costs, and the procurement of land, which are important parameters for selecting an appropriate treatment system.

Evaluation of the operational costs is based on the loading rate, the removal efficiency, the wastewater pumping cost as well as the benefit of energy saving by applying no forced aeration and low sludge production. A simplified comparison of capital and annual O&M cost, and land requirement between some of the widely used conventional wastewater treatment plant in some developing countries is summarized in Table A in Annex C1.

8.2.1 Cost Saving from the Low Energy Consumption

The energy consumption of aeration in a conventional activated sludge treatment system is generally high, i.e. approximately 1 to 1.4 kWh per kg COD removed, which is about 25% of the total operational costs (Pipyn *et al.*, 1994). In Ethiopia for example the cost of electricity is charging 70 dollar cents for a kilowatt hour (KWh). Thus, the cost saving was about 70 to 98 dollar cents per kg COD removed as there might be no energy consumption in trickling filter for forced aeration.

8.2.2 Cost Saving from the Low Sludge Production

Different researchers reported that, the operational costs of conventional domestic wastewater treatment plants in Belgium (in 1993) was about 0.63 US $ per kg COD removed. Up to 50 % of the total treatment costs go to sludge treatment. In a conventional activated sludge system, the volume of waste sludge production is supposed to be 2.5 % (v/v) according to Metcalf and Eddy (1991), moreover, the typical COD of the influent and effluent are considered to be 500 mg L^{-1} and 70 mg L^{-1}, respectively. Thus, the wet sludge production based on the COD removal is about 60 L per kg COD removed. In this trickling filter system, the sludge production in the treatment of the brewery wastewater was 1.3 % (v/v) of the treated wastewater with average COD of the influent and effluent 1286.86 mg/l and 103.98 mg/l. Based on the COD removed (chapter 1), it corresponds with 11 L sludge per kg COD removed. This represents only 50 % of the volume from a conventional activated sludge system. Therefore, a significant saving of the costs for sludge treatment can be expected. Yet there is no enough information available to compare the costs for sludge treatment based on sludge volume and concentration. Sludge treatment and disposal cost depends on the alternative methods for handling and disposing wastewater sludge. Yet, in a first approximation, it is assumed that the cost saving for one kg COD removed, by a 50 % reduction of the sludge volume could be ≤ 50 %.

Currently the different alternatives for sludge handling and disposal methods are agricultural use, composting, incineration, land filling, stabilization and solid-liquid separation. The financial estimates of the different sludge handling disposal methods are summarized in annex C2.

8.2.3 The Cost of Labor

It is difficult to evaluate the cost of labor for the small scale treatment installations. Some researchers suggested order of 0.17 US$ per kg COD removed. Indeed in a treatment system combined with a centralized organization, one skilled operator can be responsible for several plants. It must be emphasized that the current results only relate to the trickling filter setup of the present investigation. Moreover, the test-runs were operated over a period of 9 months during the investigation of the trickling filter with synthetic brewery wastewater and a period of 5 months during its operation with industrial brewery wastewater, and only at a temperature that is varying at 25°C. Hence, it is important that the delineated process approach is tested under full-scale conditions over long time periods.

However in general the above cost estimates associated with the management of sludge indicates that there is a significant cost savings achieved by the trickling filter as a result of reduced sludge volume due to the implementation of trickling filter processes for the treatment of brewery wastewater rather than the other conventional wastewater treatment processes.

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9 Summary and conclusion

In the production of one liter of beer in breweries around 3-20 L of wastewater are generated. This is heavily loaded with organic substances (BOD_5 = 1200 mg/L to 3600 mg/L, COD = 2000 mg/L to 6000 mg/L) and nutrients (N=15 mg/L to 80 mg/L, P=10 mg/L to 70 mg/L) therefore it should be treated prior to discharge to surface water bodies so as to decrease the high oxygen demand of the wastewater. Most developing countries around the globe are striving hard for a fast economic growth and associated industrialization. As a result, numerous industries are emerging mostly in the urban areas of developing countries. The wastewater, which is an obvious by product of all industries on the other hand, receives lower priority.

Breweries are a widespread industry in Africa and brewing is intrinsically a water intensive industry. According to the sectoral study and framework analysis conducted in Ethiopia, Ghana, Morocco and Uganda, water consumption and specific use (hl water/ hl beer) varies greatly between breweries in the study countries and ranges from 7.2 hl/hl in Uganda to 22 hl/hl in Ethiopia. Trickling filter is a technologically and energetically favorable opportunity for wastewater treatment for such countries. Trickling filter requires relatively low power requirements, lower operation and maintenance cost, smaller land requirement, less sludge production than suspended-growth systems and the level of skill and technical experise required is small. Therefore, this work includes a detailed study of the performance evaluation of naturally ventilated gravel-filled trickling filters for the treatment of brewery wastewater.

During the performance investigation the first operation of the trickling filter was development of bacterial bio film on the trickling filter medium. This was achieved by pumping a near neutral brewery wastewater for several days repeatedly on top of the trickling filter. The biomass development on the trickling filter medium was confirmed by monitoring influent and effluent concentration and visualizing with microscope. After the seeding of the bio reactor the acclimation behavior was studied. With respect to COD removal the acclimation period was only about 25 days with COD removal efficiency ranging from 71.5 % to 95.4 % when it is confirmed the start up period completed and the filter ready for full operation. There was only slight decrease in efficiency with increase in hydraulic and organic loading rates. The trickling filter achieved removal efficiency more than 80 % for both COD and BOD_5 for hydraulic loadings and COD loadings varying at its design load.

Investigation on the effect of pH and temperature revealed that near neutral pH (6 to 7.5) and temperature range of 25°C to 35°C is the best operating condition for both organics and nutrient removal. At its steady state the trickling filter could achieve a maximum removal efficiency of COD and BOD_5 removal (mean ± SD), 91.94 ± 2.38% and 93.10 ± 2.93% respectively at the design organic loadings of 1.5 kg COD/m^3/d and flow rate of 300 L/d.

Likewise simultaneously, reduction of nutrients ranged from 65.46 to 86.59 % and from 10.45 to 56.66 % for total nitrogen (TN) and total phosphorus (TP) respectively as the flow rates changed from 500 Ld^{-1} to 800 Ld^{-1} and at influent COD concentration of 1000 mg L^{-1} with average influent nitrogen and phosphorus concentration of 36.9 mgL^{-1} and 30.74 mg/L respectively during this investigation. The trickling filter achieved nitrogen and phosphorus removal efficiencies (mean ± SD) of 72.1 ± 18.49 % and 74.69 ± 14.14 % respectively at the design COD load and flow rate of 1648 mg L^{-1} and 800 L d^{-1}. At the steady state bio film state the trickling filter achieved a total nitrogen and total phosphorus removal efficiency of 88 % and 80 % respectively for average influent nitrogen and phosphorus concentration of nearly 39.63 and 11 mg L^{-1}. At this steady state bio film state the performance for ammonium nitrogen removal was about 98 % at influent ammonium nitrogen concentration of 22.54 mg L^{-1}. Beside this there exist strong correlation between mass loading rate and mass removal rate for both organic and nutrient removal by the bioreactor which is due to the high concentration of the sludge.

Steady state bio kinetic trickling filter models were verified using the present experimental performance data. To this end the experimental data generated during the performance evaluation of the trickling filter for removal of COD was tested using different popular trickling filter models. The experimental data had fair to excellent agreements for the models that are considered for fitting COD removal data (correlation coefficients ranging from 78 % to 96%). A model is formulated therefore for prediction of effluent property and efficiency of the bioreactor at any given operating conditions. To conclude from the present thesis, both organic substances and nutrient load of brewery wastewater can be handled in a cost-effective and environmentally friendly manner using the gravel-filled trickling filter.

The results summarized above were obtained during phase one operation of the trickling filter in which synthetic brewery wastewater was used. And in phase two operation the investigation was carried out using the local brewery wastewater. As the trickling filter was shut down for about 10 months after phase I investigation is over, start up procedure was required during phase II investigation. At this point start-up of the pilot scale trickling filter using the river stone substrate medium at ambient wastewater temperature (19 – 21 ºC) was investigated over a period of 30 days. During this time also there was no need for special inoculation, bacterial development was achieved only by feeding the raw wastewater to the filter. The acclimation period for COD, total nitrogen (TN), and total phosphorus (TP) was only about two weeks in this case. COD removal effeciency after the acclimtatization phase was 87.42 % with loading of 1.92 kg COD m^{-3} of medium d^{-1}. More than 50% nitrogen removal was first observed starting from the 4^{th} day of operation, the highest recorded value of nitrogen removal efficiency was 68.53 % and the nitrogen removal performance have stabilized by 11^{th} day of the operation. Nevertheless, the further increase in nitrogen removal is expected as the experiment is still continued. Whereas total phosphorus removal efficiency greater than 50% was observed on the 6^{th} day and the maximum TP removal efficiency was 87.5%.

The trickling filter had an average efficiency of (86.53), (95.25), (69.93) and (41.03) % for biochemical oxygen demand (COD), biological oxygen demand (BOD_5) , total nitrogen (TN) and total phosphorus (TP) respectively as the flow rates changed from 900 to 1100 L d^{-1} at influent COD concentration (mean ± SD) of 608.15 ± 59.00 mgL^{-1}. Influent nitrogen and phosphorus concentration (mean ± SD) was in the range of (10.86 ± 1.76 to 16.83 ± 4.93) and (2.44 ± 0.61 to 4.73 ± 1.80) during this operation of the trickling filter. Linear regression model revealed the existence of very high correlations between mass loading rate and mass removal rate for the elimination of all COD, nitrogen and phosphorus from the wastewater (correlations coefficients = 98 %; 99 %; 65 % and 85 % for COD, BOD_5, TP and TN respectively). From the investigation of the trickling filter at different COD loadings, average efficiencies for COD, BOD_5, TN and TP of 83.00 ± 7.04, 86.00 ± 9.31, 79.76 ± 9.14 and 51.34 ± 5.00 respectively at an organic loading of 1.35 kg COD m^{-3} medium d^{-1} and with a feed flow rate of 300 L/d was obtained. The effluent COD level of the present trickling filter is well below the provisional standards for discharge to water bodies of most developing countries (for example see Table l annex C).

The investigation carried out during phase one and phase two operations can lead to a conclusion that simultaneous organic and nutrient removal can be achieved using the mineral filled trickling filter with a high degree of both organic and nutrient reduction. Biofilm models for process obtimization and performance investigation using new biofilter media are recommended as area of future research for further improvement of nutrient removal.

10 Zusammenfassung

In Entwicklungsländern gelangen erhebliche Mengen von Brauereiabwässern direkt in Vorfluter. Unbehandelt stellen diese Abwässer eine große Belastung der Oberflächengewässer und auch des Grundwassers dar. Eine technologisch und energetisch günstige Möglichkeit zur Abwasserreinigung ist die Nutzung von Tropfkörperanlagen. Damit lassen sich zwar nicht Abscheidewirkungen wie mit Belebungsanlagen erzielen, dennoch können die Umweltbeeinträchtigungen signifikant reduziert werden. Diese Arbeit beinhaltet daher eine detaillierte Untersuchung der Möglichkeiten des Nährstoffabbaus bei der Nutzung von natürlich belüftetem Kies-gefüllten Tropfkörpern für die Behandlung von Brauereiabwasser.

Bei der Produktion von einem Liter Bier werden in Brauereien ca. 3 bis 20 L Abwasser erzeugt. Dieses ist organisch stark belastet (BSB_5 = 1200 mg/L bis 3600 mg/L, CSB = 2000 mg/L bis 6000 mg/L) und muss vor der Einleitung in den Vorfluter ausreichend mechanisch und biologisch gereinigt werden.

Bei Tropfkörperanlagen bildet sich auf dem Festkörper ein Biofilm. Darin werden die Abwasserinhaltsstoffe abgebaut. Nach Inbetriebnahme einer Labortropfkörperanlage mit Kies als Festbett (Korngröße 16 mm bis 64 mm) wurde die Entwicklung des Biofilms bei Beaufschlagung mit einem nahezu neutralen Brauereiabwasser beobachtet. Dazu dienten sowohl Messungen der CSB-Reduzierung als auch mikroskopische Untersuchungen. Die Anlaufphase dauerte 25 Tage. Danach wurden CSB- und BSB_5-Abbauraten von mehr als 80 % stabil erreicht.

Die Untersuchungen über die Wirkung von pH und Temperatur ergaben, dass nahezu neutrale pH-Werte (zwischen 6 und 7,5) und der Temperaturbereich von 25°C bis 35°C die besten Betriebsbedingungen für die Entfernung von organischer Substanz und Nährstoffen bieten. Im stationären Zustand konnte der Tropfkörper eine maximale Reinigungsleistung für CSB und BSB_5 (Mittelwert ± SD) von 91,94 ± 2,38 % und 93,10 ± 2,93 % erreichen, bei einer organischen Belastung von 1,5 kg $CSB/m^3/d$ und einem Durchfluss von 300 L d^{-1}.

Für Stickstoff und Phosphor erreichte der Tropfkörper eine Abbauleistung von (Mittelwert ± SD) 72,1 ± 18,49 % und 74,69 ± 14,14 % bei einer CSB-Belastung von 1648 mg L^{-1} und einer Durchflussmenge von 800 L d^{-1}. Im stationären Zustand erreichte der Tropfkörper eine Reinigungsleistung für Gesamtstickstoff und Gesamtphosphor von 88 % bzw. 80 % für eine mittlere Zulaufkonzentration für Stickstoff von fast 39,63 mg L^{-1} und für Phosphor von 11 mg L^{-1}.

Die experimentellen Daten ergaben gute bis hervorragende Übereinstimmungen mit den Tropfkörper-Modellen, die für die Anpassung der CSB-Daten betrachtet wurden (Korrelationskoeffizienten von 78 % bis 96 %). Während der Untersuchung des Tropfkörpers mit realem Industrieabwasser betrug die Anlaufphase für CSB, Gesamt-Stickstoff (TN) und Gesamt-Phosphor (TP) nur etwa zwei Wochen.

Der Tropfkörper erreichte danach einen durchschnittlichen Wirkungsgrad von (86,53), (95,25), (69,93) und (41,03) % für den biochemische Sauerstoffbedarf (CSB), den biologischen Sauerstoffbedarf (BSB_5), den Gesamtstickstoff (TN) und die Gesamt-Phosphor-Werte (TP), jeweils für Durchflussraten von 900 bis 1100 L d^{-1} bei Zulaufkonzentrationen des CSB (Mittelwert ± SD) von 608,15 ± 59,00 mg L^{-1}. Die Zulaufkonzentration von Stickstoff und Phosphor (Mittelwert ± SD) lag im Bereich von (10,86 ± 1,76 bis 16,83 ± 4,93) und (2,44 ± 0,61 bis 4,73 ± 1,80) mg L^{-1} während dieser Untersuchung. Bei der Untersuchung des Tropfkörpers hinsichtlich unterschiedlicher Beladungen von CSB wurden mittlere Wirkungsgrade für CSB, BSB_5, TN und TP von 83,00 ± 7,04, 86,00 ± 9,31, 79,76 ± 9,14 und 5,00 ± 51,34 % jeweils bei einer organischen Belastung von 1,35 kg CSB m^{-3} d^{-1} und mit einem Durchfluss von 300 L d^{-1} erreicht.

Die durchgeführten Untersuchungen führen zu dem Ergebnis, dass die gleichzeitige Entfernung von organischer Substanz und Nährstoffen über einen mineralgefüllten Tropfkörper einen hohen Grad der Reduzierung der organischen Substanz und der Nährstoffe erreichen kann. Für zukünftige Forschungen werden Biofilm-Modelle für die Prozessoptimierung und Performance-Untersuchungen mit neuen Bio-Filtermedien für die weitere Verbesserung der Nährstoffentfernung empfohlen.

11 Annex

11.1 Annex A Raw plotted data for figures

Table annex A1 raw plotted data for figures [3.5 & 3.6]

Time [d]	Hydraulic loading rate [$m^3/m^2/d$]	COD [mg/L]			BOD_5 [mg/L]		
		Influent	Effluent	Efficiency [%]	Influent	Effluent	Efficiency [%]
2	6.4	1247	121	90.39	1229	61.5	94.99
4		1085	113	89.59	929	46.46	94.99
5		1198	80	93.28	1017	84.5	91.69
6		1092	101	90.75	1018	50.7	95.02
7	5.6	1092	210	80.77	928	46.4	95
8		1012	208	79.45	860.20	15.5	98.8
9		1000	52	94.80	850.00	98.9	92.7
10		1000	33	96.70	850	42.5	95
11	4.8	1000	88	91.20	974	18.7	98
12		1123	107	90.47	954.6	100	89.5
14		1123	112	90.03	954.6	29.6	96.9
15		1123	120	89.31	954.6	106	88.9
16		1123	131	88.42	954.6	50.7	94.7
17	4	1247	131	89.49	954.6	47.7	95
18		1247	179	85.65	954.6	110	88.5
19		1247	177	85.81	954.6	47.73	95
20		1247	170	86.37	954.6	113	88.2

Table annex A2 raw plotted data for figure 5.2

Time [d]	Hydraulic loading rate [$m^3/m^2/d$]	Nitrogen-total [mg/L]			Phosphorus –total [mg/L]		
		Influent	Effluent	Removal efficiency	Influent	Effluent	Removal efficiency
2	6.4	25.6	6.39	75	12.5	8.22	34.24
4		40.2	8.95	77.7	28.9	15.1	47.75
5		33.4	8.95	73.2	29.4	15.1	48.64
6	5.6	65.1	17	73.9	35.8	30	16.20
7		15.8	9	43	28.9	17	41.18
8		40.2	26.8	33.3	28.9	17	41.18
9		25.9	3.94	84.8	21.3	19.12	46.65
10		25.9	2	92.3	21.3	9.26	46
11	4.8	40.2	8.92	77.81	35.6	12.5	64.89
12		40.2	5.28	86.94	35.6	19.52	45.17
14		40.2	2	95.02	35.6	16.38	53.99
15	4	40.2	8.18	79.65	35.6	16.74	52.98
16		40.2	19.44	51.64	35.6	12	66.29
17		40.2	11.51	71.37	35.6	17.73	50.19
18		40.2	15.54	61.34	35.6	31.36	11.91

Table annex A3 raw plotted data for figure 3.8

Time [d]	organic loading rate [kg COD/m^3 /d]	COD [mg/L]			BOD_5 [mg/L]		
		Influent	Effluent	Efficiency [%]	Influent	Effluent	Efficiency [%]
2	1.5	1000	154	86.72	807.29	77.5	90.40
4		1000	183	84.22	614.29	86.0	89.40
5		1000	180	84.48	811.32	98.6	87.86
6		1000	143	87.67	811.32	62.0	92.36
7		1000	87.5	92.46	---	---	---
8	2.5	1500	212	87.14	1149.00	---	---
9		1500	223	86.47	1742.00	---	---
10		1500	154	90.66	---	---	---
11		1500	183	89.00	1149.32	254	77.99
12	3.0	2000	150	93.55	1742.11	66.2	96.20
13		2000	202	91.31	1631.58	62.0	96.20
14	4.5	3000	520	83.06	1631.00	---	---
15		3000	564	81.63	1632.00	---	---

Table annex A4 raw data for figure 5.4

Time [d]	organic loading rate [kg COD/m^3 /d]	N-total [mg/L]			P-total [mg/L]		
		Influent	Effluent	Efficiency[%]	Influent	Effluent	Efficiency[%]
2	1.5	12.42	6.74	45.7	50.31	19.88	60.48
4		12.42	2.32	81.3	50.31	14.64	70.9
5		12.42	2.12	82.9	50.31	10.1	79.9
6		12.42	0.804	93.5	50.31	18.03	64.2
7		12.42	6.26	49.6	50.31	---	---
8		12.42	4.38	64.7	50.31	17.76	64.69
9	2.5	12.42	0.852	93.14	50.31	7.7	84.69
10		12.42	5.16	58.45	50.31	---	---
11		12.42	---	----	50.31	---	---
14		12.42	---	---	50.31	20.37	59.51
15	3.0	12.42	5.24	57.8	50.31	---	---
16		12.42	0.078	99.4	50.31	---	---
18	4.5	12.42	0.942	92.4	50.31	---	---
20		12.42	1.054	91.5	50.31	11.04	78.1
21		12.42	1.05	95.31	50.31	16.71	66.79

Table annex A5 Raw plotted data for figure 5.3

HLR [$m^3/m^2/d$]	Unit	Effeciency [%]			
		Nitrogen		Phosphorus	
		Average	SD	Average	SD
6.4	Mg/L	75.3	2.27	65.46	26
5.6	Mg/L	65.46	8.07	38.24	12.59
4.8	Mg/L	86.59	8.88	23.79	22.99
4	Mg/L	71.80	16.71	43.54	8.07

Table annex A 6 Plotted data for figure 5.5

COD [mg/L]	Unit	Effeciency [%]			
		Nitrogen		Phosphorus	
		Average	SD	Average	SD
1160	Mg/L	70.6	21.51	68.87	8.52
1648	Mg/L	72.1	18.49	74.69	14.14
2325	Mg/L	78.6	29.42	59.51	0
3070	Mg/L	93.07	2	72.44	8

Table annex A 7 Raw plotted data for figure 5.10 (nitrite)

Time [d]	NO_2-N [mg/L]		Time [d]	NO_2-N [mg/L]	
	1*	2*		1*	2*
1	0.018	0.031	9	0.031	0.065
2	0.027	0.026	10	0.026	0.033
3	0.032	0.060	11	0.192	0.054
4	0.032	0.060	**Max.**	**0.192**	**0.404**
5	0.013	0.047	**Min.**	**0.013**	**0.026**
6	0.013	0.046	**Ave.**	**0.049**	**0.089**
7	0.026	0.046	**SD**	**0.051**	**0.108**
8	0.026	0.404			

Table annex A 8 Raw plotted data for figure 5.10 (nitrate)

Time [d]	NO_3-N [mg/L]		Time [d]	NO_3-N [mg/L]	
	1*	2*		1*	2*
1	0.196	0.946	8	0.924	0.202
2	0.196	0.119	7	0.129	0.793
3	0.133	0.137	**Max.**	**0.924**	**0.946**
4	0.059	0.110	**Min.**	**0.059**	**0.089**
5	0.095	0.089	**Ave.**	**0.23463**	**0.31725**
6	0.145	0.142	**SD**	**0.28236**	**0.34486**

Table annex A 9 Raw plotted data for figure 5.11 (NH_4^+-N).

Time [d]	NH_4-N [mg/L]		Time [d]	NH_4-N [mg/L]	
	1*	**2***		**1***	**2***
1	22.54	1.04	12	22.54	14
2	22.54	0.99	13	22.54	13.6
3	22.54	10.9	14	22.54	0.694
4	22.54	10.9	15	22.54	0.194
5	22.54	6.78	16	22.54	0.417
6	22.54	1.24	**Max.**	**22.54**	**14**
7	22.54	7.15	**Min.**	**22.54**	**0.19**
8	22.54	11.4	**Ave.**	**22.54**	**5.56969**
9	22.54	2.45	**SD**	**0.00**	**5.08735**
10	22.54	2.8			
11	22.54	4.56			

1* = Influent, 2* = Effluent

Table annex A10 Raw plotted data for figure 5.11 (total nitrogen)

Time [d]	N-total [mg/L]		Time [d]	N-total [mg/L]	
	1*	**2***		**1***	**2***
1	39.63	13.8	12	39.63	19.2
2	39.63	13.8	13	39.63	27.4
3	39.63	11.7	14	39.63	27.8
4	39.63	24.9	15	39.63	9.05
5	39.63	26	16	39.63	2.9
6	39.63	10.4	17	39.63	4.36
7	39.63	21.1	18	39.63	6.83
8	39.63	22.7	**Max.**	**39.63**	**27.8**
9	39.63	24.1	**Min.**	**39.63**	**2.9**
10	39.63	10.1	**Ave.**	**39.63**	**15.85**
11	39.63	9.21	**SD**	**0.00**	**8.31**

1* = Influent, 2* = Effluent

Table annex A11 Raw plotted data for figure 5.11 (total phosphorus)

Time [d]	P-total [mg/L]		Time [d]	P-total [mg/L]	
	1*	**2***		**1***	**2***
1	10.92	3.50	10	10.92	6.54
2	10.92	3.36	11	10.92	5.86
3	10.92	7.16	12	10.92	2.32
4	10.92	5.28	13	10.92	2.29
5	10.92	6.03	14	10.92	1.8
9	10.92	4.36	**Ave.**	**10.92**	**4.419**
Max.	**10.92**	**7.16**	**SD**	**00.00**	**1.876**
Min.	**10.92**	**1.8**			

1* = influent , 2* = effluent

Table annex A12 Raw plotted data for figure 6.2

Time [d]	COD [mg/L]		Efficiency [%]	pH		T°	
	Inlet	Outlet		Inlet	Outlet	Inlet	Outlet
1	269	77.42	71.22	8.1	8.1	20.4	22.2
2	436	119.29	72.64	7.37	8.33	21.1	21.4
3	438	99.34	77.32	7.4	8.5	20.6	20.8
4	726	141.28	80.54	7.27	8.43	19.8	21.3
5	814	148.64	81.74	7.08	8.44	21.6	21.7
6	1555	276.32	82.23	7.30	8.16	20.5	21.5
7	1653	278.53	83.15	7.4	8.22	20.6	22.6
8	1840	306.91	83.32	7.00	7.74	21.1	22.2
9	2122	322.97	84.78	7.00	7.74	21.1	22.2
10	2142	323.66	84.89	6.2	7.62	18.5	20.2
11	2800	363.44	87.02	7.23	7.70	18.7	22.2
12	3836	482.57	87.42	7.23	7.70	18.7	22.2

Table annex A13 Raw plotted data for figure 6.2

Time [d]	Nitrogen [mg/L]		Efficiency [%]	pH		T°	
	Inlet	Outlet		Inlet	Outlet	Inlet	Outlet
2	11.4	8.49	25.54	7.37	8.33	21.1	21.4
3	18.4	13.28	27.8	7.27	8.43	19.8	21.3
4	19	8.4	55.79	7.08	8.44	21.6	21.7
5	25	10.63	57.49	7.4	8.22	20.6	22.6
7	65.4	23.86	63.52	7.00	7.74	21.1	22.2
8	77.1	27.48	64.36	6.2	7.62	18.5	20.2
9	108	39.4	67.96	6	7.84	18	22.2
11	125	39.34	68.53	6	7.19	23	23.0

Table annex A14 Raw plotted data for figure 6.2

Time [d]	Phosphorous [mg/L]		Efficiency [%]	pH		T°	
	Inlet	Outlet		Inlet	Outlet	Inlet	Outlet
3	4.73	3.92	17.12	7.27	8.43	19.8	21.3
4	6.18	3.58	42.07	7.08	8.44	21.6	21.7
5	7.11	4.17	41.35	7.4	8.22	20.6	22.6
6	13.4	6.52	51.34	7.30	8.16	20.5	21.5
9	16.9	4.91	70.95	6.2	7.62	18.5	20.2
7	23.8	2.975	87.5	7.00	7.74	21.1	22.2

Table annex A15 Raw plotted data for figures 7.1&7.2

Time [d]	Hydraulic loading rate [$m^3/m^2/d$]	COD [mg/L]			BOD_5 [mg/L]		
		Influent	Effluent	Efficiency[%]	Influent	Effluent	Efficiency[%]
1	7.2	600	96.0	84.00	535	73.2	86.32
2		612	84.2	86.24	620	78.9	87.24
3		696	83.8	87.96	541	11.3	97.91
4		674	52.7	92.18	---	---	---
5		589	51.6	91.24	---	---	---
6		554	61.33	88.93	---	---	---
7		600	51.4	91.43	---	---	---
8	7.56	601.6	336	66.36	---	---	---
9		528	177.6	68.99	524	169	67.75
10		588.8	182.6	97.72	---	---	---
11		502.7	---	---	576	---	---
12		582	78.5	86.51	576	73.2	87.29
13		619	76.2	87.69	613	50.7	91.73
14		639	91.6	85.66	---	---	---
15	8.36	520	94.6	84.81	552	---	---
16		640	85.0	86.72	552	16.9	96.94
17		600	63.2	89.47	513	22.5	95.61
18		776	35	95.49	664	45.1	93.22
19		527	43.9	91.67	---	---	---
20		560	49.6	91.14	---	---	---
21	8.76	648	229	64.66	486	---	---
22		615	193	68.62	462	220	52.38
23		615	169	72.52	462	---	---
24		616	---	---	462	56.3	87.81
25		684	220	67.84	513	135	73.68
26		587	90	84.67	439	33.8	92.30
27		646	59.0	90.87	484.5	39.4	91.87
28		648	229	64.66	486	---	---

Table annex A16 Raw plotted data for figure 7.3 & 7.4

Time [d]	Hydraulic loading rate [m^3/m^2/d]	N-total [mg/L]			P-total [mg/L]		
		Influent	Effluent	Efficiency[%]	Influent	Effluent	Efficiency[%]
2	7.2	10.9	11		3.40	1.87	45.00
4		11.6	9.70	16.38	3.12	1.84	41.03
5		9.92	8.78	11.49	2.79	1.59	43.01
6		14.2	5.13	63.87	2.60	2.54	2.3
7		9.98	4.07	59.22	---	---	---
8		8.57	4.10	52.16	---	---	---
9		10.85	6.4	41.01	---	---	---
10	7.56	25.0	32.1	---	7.30	8.83	---
11		22.8	---	---	6.29	---	---
12		22.8	20.4	10.53	6.21	6.24	---
13		19.0	---	---	3.712	---	---
14		10.5	4.73	54.95	3.21	2.60	19.00
15		12.5	3.59	71.28	3.49	2.43	43.62
16		11.1	4.17	62.25	2.90	2.29	21.03
17	8.36	9.44	5.51	41.63	2.41	2.22	7.88
18		11.3	1.89	83.24	3.05	1.37	55.08
19		8.77	2.38	72.86	2.38	1.44	39.59
20		10.5	---	---	3.24	1.41	56.48
21		6.45	1.69	73.89	1.73	1.21	30.06
22		8.48	1.86	78.07	1.88	1.31	30.32
23	8.76	23.6	12.3	47.88	3.94	2.57	34.77
24		15.5	18.1	---	3.94	3.34	15.23
25		23.6	5.76	75.59	3.94	3.18	19.29
26		15.5	4.23	72.71	3.94	2.23	43.40
27		13.9	4.41	68.27	3.21	2.40	25.23
28		15.1	6.26	58.54	---	---	---
29		10.6	3.15	70.28	2.65	1.82	31.32

Table annex A17. Plotted data for figure 7.5

HLR [$m^3/m^2/d^1$]	TN [%]	TP [%]	COD [%]	BOD_5 [%]
7.20	40.69±22.16	32.89±9.88	88.85±3.02	90.5±6.43
7.56	49.80±27.02	23.64±5.87	79.51±9.41	89.51±3.14
8.36	69.93±16.35	41.03±11.94	86.53±11.09	95.25±1.90
8.76	65.55±10.42	26.89±11.05	76.90±10.31	90.58±2.40

Table annex A18. Raw plotted data for figures 7.6 & 7.7 (mass removal rate)

HLR [$m^3m^{-2}d^{-1}$]	TN [$g/m^2/d$]	TP [$g/m^2/d$]	COD [$g/m^2/d$]	BOD_5 [$g/m^2/d$]
7.20	3.58 ± 2.28	0.77 ± 0.25	437.21 ± 39.82	406.74 ± 34.13
7.56	7.17 ± 3.26	0.91 ± 0.31	552.49 ± 96.21	636.00 ± 49.93
8.36	12.38 ±4.19	2.29 ± 1.11	1076.60 ± 216.9	1090.76 ± 129.21
8.76	26.35 ± 8.71	2.25 ± 1.05	1248.73 ± 239.5	1000 ± 54.68

Table annex A19. Raw plotted data for figures 7.6 & 7.7 (mass loading rate)

HLR [$m^3/m^2/d$]	TN [$g/m^2/d$]	TP [$g/m^2/d$]	COD [$g/m^2/d$]	BOD_5 [$g/m^2/d$]
7.20	8.64 ±1.54	2.32 ±0.19	491.92 ±39.60	450.11 ± 37.77
7.56	16.99 ± 6.90	3.82±0.36	692.04±57	709.98±30.93
8.36	17.69 ± 3.48	5.38±1.71	1244.36 ±173.55	1146.83±155.46
8.76	40.73 ± 12.80	8.45±1.4	1503.34±88.32	1103.11±54.34

Table annexes A 20 Raw plotted data for figure 7.9

Time [d]	Organic loading rate [kg COD/m^3/d]	COD [mg/L]			BOD_5 [mg/L]		
		Influent	Effluent	Efficiency [%]	Influent	Effluent	Efficiency [%]
2	1.35	906	155	82.89	5.7	62	87.77
4		889	265	70.19	507	124	75.54
5		910	83	90.88	593	39.4	93.36
6		882	109	87.64	621	67.6	89.11
7		934	158	83.08	---	---	---
8		933	156	83.28	---	---	---
7	1.1	676	165	75.59	414	141	65.94
8		700	228	67.43	---	---	---
9		785	362	50.45	---	---	---
10		640	204	68.13	518	---	---
11		700	134	80.86	625	22.5	96.4
12		713	54	92.43	625	197	68.48
13		779	189	75.74	693	152	78.07

Table annex A 21 Raw plotted data for figure 7.9

Time [d]	Organic loading rate [kg COD/m^3/d]	N-total [mg/L]			P-total [mg/L]		
		Influent	Effluent	Efficiency [%]	Influent	Effluent	Efficiency [%]
2	1.35	11.4	2.06	81.93	3.00	1.51	49.67
4		12.0	2.1	82.50	3.03	1.37	54.79
5		12	2.34	80.5	2.99	1.35	54.85
6		12	2.88	76.00	2.91	1.43	50.86
7		14.5	1.05	92.76	3.19	1.42	55.49
8		8.6	3.02	64.88	3.02	1.74	42.38
9	1.1	8.99	4.02	55.28	2.29	1.90	17.03
10		11.4	3.00	73.68	2.43	1.66	31.69
11		9.09	3.73	58.97	2.57	1.61	37.35
12		58.5	13.2	77.44	10.5	6.97	33.62
13		88	23.4	73.41	15.2	12.4	18.42
14		11.7	2.27	80.59	3.58	2.41	32.68
15		12.5	3.61	71.12	3.63	2.36	34.99
16		13.6	4.99	63.31	3.55	2.39	32.68

11.2 Annex B cost of wastewater treatment plants

Table annex B 1 Summary of capital and annual O&M cost, and land requirement for UASB, WSP, and ASP of some developing countries.

Process	Treatment volume	Unit	Capital cost[a]	Unit	Land requirment	Unit	Annual O&M	Unit	Country	Reference	Remarks
UASB	36,000	m^3/d	441	US\$/m^3/d	14	m^2/m^3/d	20	US\$/m3/d	India	Tare et al. (2003)	Tannery effluent composed. US\$1 = 48.27 Rs. (2002/03)
UASB +Pond	20,000–400,000	m^3/d	34.7–45.6	US\$/m^3/d	1.70–1.98	m^2/m^3/d			India	Binnie Thames Water (1996)	
UASB+pond			68.5–85.6	US\$/m^3/d	1.1–1.7	m^2/m^3/d			India	Arceivala (1998)	US\$1 = 32.427 Rs. (ave. 1995)
UASB	50,000	PE	17.8	US\$/PE	0.12	m^2/PE	0.53	US\$/PE	Egypt	Schellinkhout (1993)	Capital cost includes land. US\$1 = 3.37 LE (December 1993)
UASB+pond	50,000	PE	27.9	US\$/PE	0.64	m^2/PE	0.53	US\$/PE	Egypt	Schellinkhout (1993)	Capital cost includes land. US\$1 = 3.37 LE (December 1993)
UASB+trickling filter	50,000	PE	31.5	US\$/PE	0.22	m^2/PE	0.71	US\$/PE	Egypt	Schellinkhout (1993)	Capital cost includes land. US\$1 = 3.37 LE (Dec 1993)
WSP	30,000	m^3/d	167	US\$/m^3/d	15.3	m^2/PE	1.67	US\$/m3/d	Yemen	Arthur (1983)	

WSP	50,000	PE	35.6	US$/PE	1.7	m^2/PE	0.53	US$/PE	Egypt	Schellinkhout (1993)	Capital cost includes land. US$1 = 3.37 LE (Dec 1993)
WSP	20,000–400,000	m^3/d	12.4–18.0	US$/$m^3$/d	12.5–14.0	m^2/m^3/d			India	Binnie Thames Water (1996)	
WSP			25.7–34.3	US$/$m^3$/d	5.6–15.6	m^2/m^3/d			India	Arceivala (1998)	US$1 =32.427 Rs (ave. 1995)
ASP	2150	m^3/d	186	US$/$m^3$/d	9.5	m^2/m^3/d	47	US$/m3/d	India	Tare et al. (2003)	Tannery effluent composed. US$1 = 48.27 Rs (2002/03)
ASP	20,000–400,000	m^3/d	50.0–60.8	US$/$m^3$/d	0.73–1.01	m^2/m^3/d			India	Binnie Thames Water (1996)	
ASP			102.8–119.9	US$/$m^3$/d	1.1–1.4	m^2/m^3/d			India	Arceivala (1998)	US$1 = 32.427 Rs (ave. 1995)
ASP			$638*Q^{0.219}$	US$/$m^3$/d					China	Li (1987)	
ASP					$34.3*Q^{-0.33}$	m^2/m^3/d			China	Li et al. (1990)	
ASP	120,000 - 540,000	PE			$4.05*PE^{-0.228}$	m^2/PE			Greece	Tsagarakis et al. (2003)	
ASP	40,000 - 180,000	PE	$159.4*PE^{-0.046}$	US$/PE					Greece	Tsagarakis et al. (2003)	
ASP	40,000 - 540,000	PE					$212*PE^{-0.328}$	US$/PE	Greece	Tsagarakis et al. (2003)	
ASP	>= 10,000	m^3/d	$49,630*Q^{-0.277}$	US$/$m^3$/d			$578*Q^{-0.19}$	US$/m3/	Japan	JSWA (2001)	US$1 =127.36 JP yen

								d			(December 2001)
ASP	10,000 - 500,000	m^3/d			$212*Q^{-0.514}$	m^2/m^3/d			Japan	JSWA (1999)	US$1 = 102.68 JP yen (December 1999)
ASP	70,000 - 650,000	PE					$313*PE^{-0.3}$	US $/PE	Sweden	Balmer and Mattsson (1994)	US$1 = 8 SEK (July 1993)

Q = treatment volume, PE = population equivalent.
[a]Capital cost does not include land cost unless it mentions in the remarks

Annex B 2 The financial estimates of the different sludge handling disposal methods

Agriculture use

When making a financial estimate of sludge spread on farmland, the costs that have to be taken in considerations are transport costs from treatment plant to storage, storage investments and operating costs, transport costs from storage to farmer, expenses for analysis of sludge quality, expenses for analysis of soil quality, administrative expenses for e.g. declaration of sludge, conclusion of agreements with farmers and control of application. The price for agricultural use of sludge is in the order of DEM 150-400/ton of sludge with 20% dry solids (European Environment Agency, 1997).Therefore the cost saving by the trickling filter per kg of COD removed can be estimated to be in the order of DEM 75-200 according to the above calculations. It should be noted, however, that the price depends on local conditions and may differ considerably from the above.

Composting

The following costs must be taken into account when a sewage water treatment plant decides to compost its sludge :the cost of transporting the sludge to the composting plant, capital investment and engineering costs for the composting system itself and the plant infrastructure (buildings, aeration, odor and air cleaning equipment, machinery for turning and mixing the compost, sieves, conveyers if necessary, front loaders), plant operating costs: personnel, energy (electricity, fuel), bulking agents (including the cost of transporting bulking agent to the composting plant if necessary), maintenance, overhead expenses, taxes, quality control expenses:characterization of wastes and the compost end product as well as process evaluation marketing costs, including market studies and marketing materials, cost of transporting the compost from the composting plant (when necessary).An indication of price range for composting is between DEM 275-525/tonne in France for example. Again according to the above calculations made for the trickling filter, the cost saving per k.g of COD removed can be estimated to be in the order of DEM 137-262 for the price in France for example.

Drying

The disadvantage here is the high investment cost relative to other methods. No reliable and generally valid estimates are known since local factors influence heavily. A few questions may clarify this point. It may for instance be questioned whether making biogas is favorable or not. What is the price per kWh as compared with the market price for electric power?, how much will digestion of sludge reduce the organic content and its value as fuel and fertilising/soil conditioning?, the heat supplied to the dryer, for instance through steam, is still present, but at a lower temperature. Is there a need for this low temperature heat, and how should it be priced?, does the sludge owner have only one 'customer', except for deposit?, how will this one and only customer pay him in the future, and what changes in the deposit cost are expected?

For example a plant without digestion handling 2,400 tons of DS/year in the form of 12,000 tons of dewatered sludge (20% DS) a year, may have a cost of approx. NOK 1,300/t of DS (approx. DEM 295, FFR 1,000, GBP 130) including capital (investment NOK 5-6 mill.) and operating cost (lime, polymer, transport, etc.).If the cost of the building, storage, required heat and disposal of the dry material is included as well, the cost may typically be in the order of DEM 600 to 700 per ton of DS. That implies according to the sludge volume calculated for the trickling filter there is a cost saving of still about 50% of what is needed for handling the sludge using the drying method.

Incineration

The following costs have to be considered in the decision making process for treatment of sludge by incineration: cost of storage systems necessary, cost of furnace, treatment of off-gas and other incineration residues, i.e. bottom ash, fly ash, clinker, other peripheral costs either for existing plants or in the case of new plants, fixed, proportional operating costs: personnel, consumables (i.e.fuel, electricity and chemicals for flue gas cleaning), maintenance, taxes, etc.,cost of transporting the sludge to the treatment site, cost of quality control (raw sludge and sub-products),marketing costs generated by the recycling of some sub-products. And this also implies that the significant reductions in sludge volume achieved by the trickling filter when compared to the conventional activated sludge process has impact on all the expected cost of incineration which are described above.

Land filling

The specific price of establishing a sludge deposit (including the cost of geological and hydro geological investigations, area, liner (plastic or clay), leachate and gas collection and treatment systems (drainage systems), fences, planting, soil for covering the sludge, monitoring wells, control and analysis of leachate, weigh bridge, buildings, machines, pumps etc., but excluding the price of the land) is approx. HUF 10,000 or USD 67, or DEM 100 per square meter (1996 price level). There are huge differences between the Austrian and German specific prices with regard to the disposal cost of 1 ton of waste. The disposal cost in Styria (Austria) varies between 800-3,500 ATS (USD 80-350). In Germany the price was in 1991 DEM 300 - 600, showing a constant increase over the past few years. As an order of magnitude, specific treatment costs may vary as indicated in table 9.1.

Table annex B2 Sludge treatment costs in DEM per ton of dry substance

Utilization in agriculture/forestry	150-400
Composting	250-600
Drying	300-800
Incineration*	450-800
Land filling	200-600
*Lower commercial prices possible in case of paying marginal prices for sludge incineration at waste incineration plants	

11.3 Annex C Provisional standards for industrial pollution control in Ethiopia

INTRODUCTION

Industrial development can be made compatible with environmental conservation. Hence, industrial pollution and resource degradation need not arise if a framework of sustainable development is appropriately formulated and implemented. Failure to halt further deterioration of environmental quality arising from industrial pollution may jeopardize the health of a large segment of the population with serious political and socio-economic consequences.

The government of the Federal Democratic Republic of Ethiopia has placed a high premium on the environment. It has established the Environmental Protection Authority (EPA) by the proclamation no 9/1995 with statutory responsibility for overall protection of the environment. The Environmental Policy of Ethiopia was formulated and approved by the government in April 1997.

Implementation of the Policy is the next task that needs to be undertaken. Introducing these standards is part of the implementation of the Policy and the environmental pollution abatement strategy contained therein.

Environmental protection measures are only meaningful if the environment to be protected is adequately understood. Neither overprotection nor under protection of the environment is desirable. Ideally, standards are set based on country specific baseline data and information, which are scanty in the present circumstances. An alternative approach is to adapt the standards of developing countries having similar socio-economic, technological, and climatic conditions.

In the preparation of this document, standards from the following developing countries have been consulted; Bangladesh, Pakistan, India, Jamaica, China, Thailand, Uganda, Nigeria, Zambia, and Kenya. Information was also obtained from development agencies such as The World Bank, United Nations Environment programme (UNEP), United Nations Industrial Development Organization (UNIDO), and from other information sources such as the European Union and the United States Environmental Protection Agency.

Where the standards were deemed relevant and appropriate for Ethiopian conditions they have been adopted, where deemed inappropriate they have been modified on the basis of practical experience.

The fact that the majority of people in Ethiopia use the receiving water bodies for drinking, washing and bathing were also considered.

These standards are being introduced to be used throughout the country subject to amendment as more information on the state of pollution is made available. The regional states can establish more stringent standards taking into consideration particular ecological conditions in their localities provided that these present standards are used as the minimum.

The purpose of introducing the standards is to prevent significant industrial pollution by indicating standards which must be observed and by indicating pollution limits beyond which the environment would not tolerate.

Limit values for emissions to water shall be interpreted in the following way:-

During continuous monitoring:

a) No flow value shall exceed the specified limit.

b) No pH value shall deviate from the specified range.

c) No temperature value shall exceed the limit value.

During Non-Continuous Monitoring:

d) No pH value shall deviate from the specified range.

e) No temperature value shall exceed the limit value.

f) For parameters other than pH, temperature and discharge, eight out of ten consecutive results, calculated as daily mean concentration or mass emission values on the basis of flow proportional composite sampling, shall not exceed the emission limit value. No individual result similarly calculated shall exceed 1.2 times the emission limit value.

g) For parameters other than pH, temperature, and flow, no grab sample value shall exceed 1.2 times the emission limit value.

The daily raw waste load is defined as the average daily mass arising for treatment over any three-month period. Calculations of the removal rates should be based on the differences between the waste loads entering the treatment plant and those discharged following treatment to the receiving water. The amounts removed by treatment (chemical, physical, biological) may be included in the calculation.

Standard for MALTING, BREWING, DISTILING, PRODUCTION OF WINES AND OTHER ALCOHOLIC LIQUOURS

Table annex C Limit Values for Discharges to Water

Parameter	**Limit Value**
Temperature	40 °C
pH	6 – 9
BOD_5 at 20°C	90% removal or 60 mg/l, whichever is less
COD	90% removal or 250 mg/l, whichever is less
Suspended solids	50 mg/l
Total ammonia (as N)	20 mg/l
Total nitrogen (as N)	80% removal or 40 mg/l, whichever is less
Total phosphorus (as P)	80% removal or 5 mg/l, whichever is less
Oils, fats, and grease	15 mg/l
Mineral oils at the oil trap or interceptor	20mg/l

12 Theses / Thesen zur Dissertation

Simultaneous Organic and Nutrients Removal in a mineral filled pilot scale trickling filter treating brewery wastewater

Presented by

Haimanot Habte Lemji

I Problem and rationale for research

During the production of beer huge wastewater generation from cooling (eg. saccharification cooling, fermentation) and washing units often causes several environmental problems. Substantial improvements has been made in the past, however it has been estimated that approximately 3 to 22 L of waste effluent is generated per liter of beer produced in breweries. Brewing effluent contains high organic contents and nutrients. The removal of organic compounds from the wastewater is important to avoid anaerobic conditions in the receiving waters. Nutrients like nitrogen (N) and phosphorous (P) should also be removed to avoid algal blooms that disturb the ecosystem of the receiving waters. Furthermore turbidity and color reduces the penetration of light, which, in turn, affects photosynthesis, thereby affecting the primary link in the food chain.

The large amount of energy input in wastewater treatment, especially energy consumed by aeration procedures, in aerobic treatment, has been considered as a challenge for many years in wastewater treatment in many countries. For instance in the brewery industry at Paonta Sahib (H.P), India activated sludge process initially used in 1997 is suffering from high energy requirements for the aeration and inconsistency in achieving the effluent standards. And a number of pollution related studies have confirmed that about 90% of industries including breweries in the capital city of Ethiopia for instance are simply discharging their effluent into nearby water bodies, streams and open land without any form of treatment due to the high cost of pollution control technologies. On the other hand the feasibility of some of these treatment technologies is limited by sludge handling problem, for example activated sludge normally exhibits poor settleability, as such, and fixed film systems that would involve trickling filters, rotating biological contactors, et cetera are recommended.

Trickling bio filters and tower filtration technologies are regarded as well-established treatment technologies for wastewater treatment. Unlike the other conventional treatment technologies, trickling filter do not require high investment in mechanical or energy demanding equipment and does not require much human attendance for operation and maintenance of the systems. Therefore in this study, the detailed investigation aimed at analyzing the performance of gravel-filled, naturally aerated trickling filter on brewing industry wastewater.

II Methodological Approach

A pilot scale trickling filter consisted of a plexiglas tube with an inner diameter of 40 cm and a total height of 180 cm used as experimental plant. Sampling ports are located at fixed intervals of 260 mm along the height of the bio filter. At the top of the filter, a fixed flow distributor was installed to facilitate a uniform distribution of the wastewater fed to the filter's free surface. The wastewater was fed to the reactor after being homogenized in a wastewater reservoir. Also, a secondary clarifier was installed to collect and settle the effluent from the filter's draining system. Locally available stone gravel was filled inside the trickling filter tank that served as a support for the growth of the bacteria. Air is driven through the trickling filter by vertical pressure differences developed by thermal buoyancy. The warm air inside the bioreactor is less dense than cooler air outside, and thus will try to escape from openings high up in the trickling filter column; cooler denser air will enter openings lower down. The process will continue if the air entering the bioreactor is continuously heated, typically by casual or solar gains therefore there was no external aeration of the bioreactor. The performance evaluation of the bioreactor conducted in two phases in this thesis. In the first phase the investigation was performed using synthetic brewery wastewater and in the second phase the performance evaluation was conducted using real industrial brewery wastewater.

The COD and major nutrient concentrations of the wastewater (Nitrite – N, Nitrate – N, Ammonium–N, total nitrogen and total phosphorous) were determined spectrophotometrically. In all the cases concentration is determined by the addition of a reagent to the raw water sample. After allowing time for color development, the color is read at the wavelength between 400-500 nm. Where as BOD_5 measurment was carried out using oxi top instrument after sample incubation. Biomass concentrations were determined by weighing dried (24 h, 105 °C) 100 ml samples of the liquid phase. During the operation of the trickling filter, control of excess sludge was achieved by washing with diluted sodium hydroxide and when indicated by starvation.

III Main results and new evidence

During phase one investigation (operation of the trickling filter using synthetic brewery wastewater), the bioreactor achieved removal efficiency above 80 % for both COD and BOD_5 at waste water flow rate range of 500 Ld^{-1} to 800 Ld^{-1} including recirculation flow and at influent COD concentration of about 1000 mg L^{-1}. Likewise efficiency of removal for nutrients ranged from 65.46 % to 86.595 % and from 10.45% to 56.66 % for total nitrogen (TN) and total phosphorus (TP) respectively at the given flow rate and COD loadings .The influent nitrogen and phosphorous concentration was 36.9 mgL^{-1} and

30.74 mg L^{-1} respectively during this operation of the trickling filter. During phase two investigation (operation of the trickling filter using real industrial brewery wastewater), the trickling filter achieved average efficiencies of (86.53), (95.25), (69.93) and (41.03) % for biochemical oxygen demand (COD), biological oxygen demand (BOD_5), total nitrogen (TN) and total phosphorus (TP) respectively as the flow rates changed from 900 to 1100 L d^{-1} and at influent COD concentration (mean ±SD) of 608.15 ± 59.00 mgL^{-1}. Influent nitrogen and phosphorus concentration (mean±SD) was in the range of (10.86±1.76 to 16.83±4.93) and (2.44±0.61 to 4.73±1.80) during this operation of the trickling filter. Therefore as new evidence from the present investigation, brewery effluent can be treated in a cost effective and environmentally friendly manner using the mineral filled trickling filter.

IV Conclusions and outlook

Trickling filter can be employed to achieve excellent effluent quality with respect to COD load and moderate effluent quality with respect to nutrient load in the treatment of brewery wastewater. Savings as a result of low land requirements (because smaller foot print), low energy demand, reduced sludge management cost and lower maintanance costs can be gained if trickling filter is used instead of other conventional treatment processes. From this point of view trickling filter can be proposed to low income countries that are highly challenged by high cost of wastewater treatment plant.

13 CURRICULUM VITAE

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14 Declaration of primary authorship

Here by I declare that I independently written the present thesis for doctorate and without the help of others. This thesis and the work presented in it is entirely my own. Where I have consulted the work of others, this is always clearly stated. The present work is an original work and not submitted either in Germany or elsewhere for examination in the present or similar version.