DIFFUSER FOULING MITIGATION, WASTEWATER CHARACTERISTICS AND TREATMENT TECHNOLOGY IMPACT ON AERATION EFFICIENCY

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ABSTRACT

Achieving energy neutrality has shifted focus towards aeration systems optimization, due to the high energy consumption of aeration processes in modern advanced wastewater treatment plants. The activated sludge wastewater treatment process is dependent on aeration efficiency which supplies the oxygen needed in the treatment process. The process is a complex heterogeneous mixture of microorganisms, bacteria, particles, colloids, natural organic matter, polymers and cations with varying densities, shapes and sizes. These activated sludge parameters have different impacts on aeration efficiency defined by the OTE, % and alpha.

Oxygen transfer efficiency (OTE) is the mass of oxygen transferred into the liquid from the mass of air or oxygen supplied, and is expressed as a percentage (%). OTE is the actual operating efficiency of an aeration system. The alpha Factor (α) is the ratio of standard oxygen transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency of clean water (SOTE). It is also referred to as the ratio of process water volumetric mass transfer coefficient to clean water volumetric mass transfer coefficient. The alpha factor accounts for wastewater contaminants (i.e. soap and detergent) which have an adverse effect on oxygen transfer efficiency. Understanding their different impacts and how different treatment technologies affect aeration efficiency will help to optimize and improve aeration efficiency so as to reduce plant operating costs.

A pilot scale study of fine pore diffuser fouling and mitigation, quantified by dynamic wet pressure (DWP), oxygen transfer efficiency and alpha measurement were performed at Blue Plains, Washington DC. In the study a mechanical cleaning method, reverse flexing (RF), was used to treat two diffusers (RF1, RF2) to mitigate fouling, while two diffusers were kept as a control with no reverse flexing. A 45 % increase in DWP of the control diffuser after 17 month of operation was observed, an indication of fouling. RF treated diffusers (RF1 and RF2) did not show any significant increase in DWP, and in comparison to the control diffuser prevented a 35 % increase in DWP. Hence, the RF fouling mitigation technique potentially saved blower energy consumption by reducing the pressure burden on the air blower and the blower energy requirement. However, no significant impact of the RF fouling mitigation treatment technique in preventing a decrease in alpha-fouling (αF) of the fine pore diffusers over time of operation was observed. This was because either the RF treatment method maintained wide pore openings after cleaning over time, or a dominant effect of other wastewater characteristics such as the surfactant concentration or particulate COD could have interfered with OTE.

Further studies on the impact of wastewater characteristics (i.e., surfactants and particulate COD) and operating conditions on OTE and alpha were carried out in another series of pilot and batch scale tests. In this study, the influence of different wastewater matrices (treatment phases) on oxygen transfer efficiency (OTE) and alpha using full-scale studies at the Blue Plains Treatment Plant was investigated. A strong relationship between the wastewater matrices with oxygen transfer characteristics was established, and as expected increased alphas were observed for the cleanest wastewater matrices (i.e., with highest effluent quality). There was a 46 % increase in alpha as the total COD and surfactant concentrations decreased from 303 to 24 mgCOD/L and 12 to 0.3 mg/L measured as sodium dodecyl sulphate (SDS) in the nitrification/denitrification effluent with respect to the raw influent. The alpha improvement with respect to the decrease in COD and surfactant concentration suggested the impact of one or more of the wastewater characteristics on OTE and alpha.

Batch testing conducted to characterize the mechanistic impact of the wastewater contaminants present in the different wastewater matrices found that the major contaminants influencing OTE and alpha were surfactants and particulate/colloidal material. The volumetric mass transfer coefficient (k_La) measurements from the test also identified surfactant and colloidal COD as the major wastewater contaminants present in the influent and chemically enhanced primary treatment (CEPT) effluent wastewaters impacting OTE and alpha. Soluble COD was observed to potentially improve OTE and alpha due to its contribution in enhancing the oxygen uptake rate (OUR). Although the indirect positive impact of OUR on alpha observed in this study contradicts some other studies, it shows the need for further investigation of OUR impacts on oxygen transfer. Importantly, the mechanistic characterization and quantitative correlation between wastewater contaminants and aeration efficiency found in this study will help to minimize overdesign with respect to aeration system specification, energy wastage, and hence the cost of operation. This study therefore shows new tools as well as the identification of critical factors impacting OTE and alpha in addition to diffuser fouling.

Gas transfer depression caused by surfactants when they accumulate at the gas-liquid interface during the activated sludge wastewater treatment process reduces oxygen mass transfer rates, OTE and alpha which increases energy cost. In order to address the adverse effect of surfactants on OTE and alpha, another study was designed to evaluate 4 different wastewater secondary treatment strategies/technologies that enhances surfactant removal through enhanced biosorption and biodegradation, and to also determine their effect on oxygen transfer and alpha. A series of pilot

and batch scale tests were conducted to compare and correlate surfactant removal efficiency and alpha for a) conventional high-rate activated sludge (HRAS), b) optimized HRAS with contactor-stabilization technology (HRAS-CS), c) optimized HRAS bioaugmented (Bioaug) with nitrification sludge (Nit S) and d) optimized bioaugmented HRAS with an anaerobic selector phase technology (An-S) reactor system configuration. The treatment technologies showed surfactant percentage removals of 37, 45, 61 and 87 %, and alphas of 0.37 ±0.01, 0.42 ±0.02, 0.44 ±0.01 and 0.60 ±0.02 for conventional HRAS, HRAS-CS, Bioaug and the An-S reactor system configuration, respectively. The optimized bioaugmented anaerobic selector phase technology showed the highest increased surfactant removal (135 %) through enhanced surfactant biosorption and biodegradation under anaerobic conditions, which also complemented the highest increased alpha (62 %) achieved when compared to the conventional HRAS. This study showed that the optimized bioaugmented anaerobic selector phase reactor system configuration is a promising technology or strategy to minimize the surfactant effects on alpha during the secondary aeration treatment stage.

Diffuser Fouling Mitigation, Wastewater Characteristics and Treatment Technology Impact on Aeration

Victory Oghenerabome Odize

GENERAL AUDIENCE ABSTRACT

In the activated sludge process, the energy requirement for aeration which also includes nitrogen removal is a major operating expense for utilities, and it has limited the ability of most water and wastewater reclamation facilities to achieve energy neutrality. Aeration has therefore become one of the most energy and capital intensive aspects of wastewater treatment. There are still knowledge gaps and mechanistic understanding of the impact of wastewater characteristics and treatment processes on aeration efficiency, which past and current studies are yet to provide. Aeration efficiency is defined by oxygen transfer efficiency and alpha (an indicator of wastewater contaminant effect on aeration efficiency). This study provided an insight into important wastewater characteristics, treatment processes and operational parameters contributing to aeration cost. An understanding of the impacts of wastewater characteristics and how different treatment technologies affect aeration efficiency as discussed in this study will help design engineers and operators to optimize and improve aeration efficiency, so as to reduce plant operating costs.

The first study objective on fine bubble diffuser fouling dynamics and physical treatment method quantified by dynamic wet pressure (DWP), oxygen transfer efficiency and alpha measurement was carried out in a pilot reactor. DWP quantified the fouling dynamics of fine pore diffusers. A diffuser fouling physical treatment (reverse flexing, RF) method was able to mitigate fouling of the fine pore diffusers by preventing an increase in DWP normally observed in fouled fine pore diffusers. The RF treatment method reduced fouling by 35 % as compared to the control diffuser

(without reverse flexing). This will reduce the pressure burden and air blower energy requirement.

The second study objective evaluated the impact of different wastewater characteristics and removal in different stages on aeration efficiency. Test results in this study showed that surfactant and particulate COD fractions were the major characteristics constituents contained in wastewater that depressed aeration efficiency defined by OTE and alpha. Soluble COD did not show any inhibiting effect on OTE and alpha.

The third study objective evaluated three different optimized wastewater treatment technologies of surfactant removal during aeration treatment process; 1) High rate activated sludge (HRAS) with contactor-stabilization technology (The contactor stabilization process) (HRAS-CS); 2) HRAS bioaugmented (BioAug) with nitrification sludge (Nit S); and 3) Bioaugmented HRAS with an anaerobic selector phase (An-S) configuration. All three technologies increased surfactant removal through enhanced biosorption and biodegradation to various degrees when compared the conventional high rate activated sludge treatment, but the *An-S* treatment technology achieved the highest surfactant removal and alpha improvement. The study also established the optimum performance process conditions for each optimized treatment technology.

DEDICATION

This Dissertation is dedicated to GOD Almighty WHOSE Grace alone saw me through, to my wife Sonia Odize, my sons Jaden and David Odize, and also to my Dad Engr Daniel Odize and the entire Odize family. Without their support, I could never have dreamt nor completed the pursuit of the Ph.D. degree.

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ATTRIBUTION

Each co-author is duly credited for his or her contribution to this work, both in their sharing of ideas and technical expertise.

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LIST OF ACRONYMS

A	Surface Area
ASP	Activated Sludge Process
AE	Aeration Efficiency
ANNAMOX	Anaerobic Ammonia Oxidation
AN-S	Anaerobic Selector
AOB	Ammonia Oxidizing Bacteria
AWTP	Advanced Wastewater Treatment Plant
BNR	Biological Nutrient Removal
BioAug	Bioaugmented High Rated Activated Sludge
C	Effective Average DO Concentration In The Liquid Phase
CAS	Conventional Activated Sludge
cCOD	Colloidal COD
CEPT	Chemically Enhanced Primary Treatment
C_∞^*	Saturated Dissolved Oxygen Concentration
CMAS	Completely Mixed Activated Sludge
CO_2	Carbon Dioxide
COD	Chemical Oxygen Demand
CS	Contact-Stabilization
CSTR	Continuously Stirred Tank Reactor

Den	Denitrification
dC	Change In Dissolved Oxygen Concentration
dt	Change In Time
DO	Dissolved Oxygen
DWP	Dynamic Wet Pressure
Effl	Effluent
ENR	Enhanced Nutrient Removal
EPDM	Ethylene-Propylene-Diene Monomer
EP	Energy Production
EPS	Extracellular Polymeric Substance
EPS_C	EPS in Contactor
EPS_S	EPS in Stabilizer
F	Fouling Factor
ffCOD	Flocculated-Filtered COD
α F	Alpha Fouling Factor
F/M	Food to Microorganism ratio
HRAS	High Rate Activated Sludge
HRT	Hydraulic Retention Time
K_La	Volumetric Oxygen Mass Transfer Rate
αK_La	Volumetric Oxygen Mass Transfer Under Process Condition
LB-EPS	Loosely Bounded-EPS
MBAS	Methylene Blue Active Substances

MBRs	Membrane Bioreactors
MGD	Million Gallons per Day
MLSS	Mixed Liquor Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
MP	Membrane Panel Diffusers
N_2	Di-Nitrogen gas
Nit	Nitrification
Nit S	Nitrification Sludge
NOB	Nitrite Oxidizing Bacteria
O_2	Oxygen
OTE	Oxygen Transfer Efficiency
OTR	Oxygen Transfer Rate
OUR	Oxygen Uptake Rate
pCOD	Particulate COD
pH	Potential for Hydrogen
Rem	Removal
RAS	Return Activated Sludge
RF	Reverse Flexing
rbCOD	Readily Biodegradable COD
sbCOD	Slowly Biodegradable COD
SAE	Standard Aeration Efficiency
SOTE αSOTE	Standardized Oxygen Transfer Efficiency of Clean Water

	Standardized Oxygen Transfer Efficiency of Process Water
SCOD	Soluble COD
SRT	Solids Retention Time
SOTR	Standardized Oxygen Transfer Rate
SDS	Sodium Dodecyl Sulphate
SMP	Soluble Microbial Product
SVI	Sludge Volume Index
T	Temperature
tCOD	Total COD
TB-EPS	Tightly Bounded-EPS
TSS	Total Suspended Solids
US EPA	United States Environmental Protection Agency
V	Volume of Aeration Tank
VSS	Volatile Suspended Solids
WAS	Waste Activated Sludge
WRRF	Water Resource Recovery Facility
w.wt	Wastewater
WWTP	Wastewater Treatment Plant

CHAPTER ONE

INTRODUCTION

Figure 0.1: Aerial photograph of the Blue Plains Advanced Wastewater treatment plant where the experimental work presented in this thesis was conducted.

1.1 BACKGROUND ON THE PROBLEM

In developed countries, aerobic biological wastewater treatment is a major treatment process. The activated sludge process, though energy and capital intensive, allows for flexibility with regard to plant layout, equipment selection, reactor design and operational control to minimize cost and still meet stipulated effluent discharge limits (Reardon, 1995; Houck and Boon, 1981; Rosso and Stenstrom, 2005; EPRI and WERF, 2013). The activated sludge process (ASP) is dependent on aeration, which supplies the oxygen needed by the microbes in the treatment process. The energy consumption of aeration processes in modern advanced wastewater treatment plants is now about 49 to 80% of the plant total energy requirement (Fig 1.2), making aeration the most energy and capital intensive aspect of wastewater treatment (MOP32, 2009; Houck and Boon, 1981; Rosso and Stenstrom, 2005). In the US as of 1996, the annual electricity use by municipal wastewater treatment plants was 17.4 billion kWh/yr, and in the year 2013 it rose to 30.2 billion kWh/yr, a 74% increase (EPRI and WERF, 2013). The huge jump in energy consumption is a consequence of the addition of ammonia and nitrogen removal processes during wastewater treatment, which today is becoming the minimum standard for effluent discharge. Therefore, existing and new wastewater treatment infrastructures have to add nitrification units with larger aeration tanks and longer sludge retention times, leading to an additional energy demand for oxygen transfer needed in the nitrification process. This increase in power and energy cost create an incentive for the replacement of low efficiency aeration systems (i.e., coarse bubble, poor efficient blowers) with new and low cost highly efficient ones (i.e., fine pore diffusers, high efficient blowers) (see Table 1) having new technological configurations (Rosso et al., 2008; Stenstrom et al., 1983).

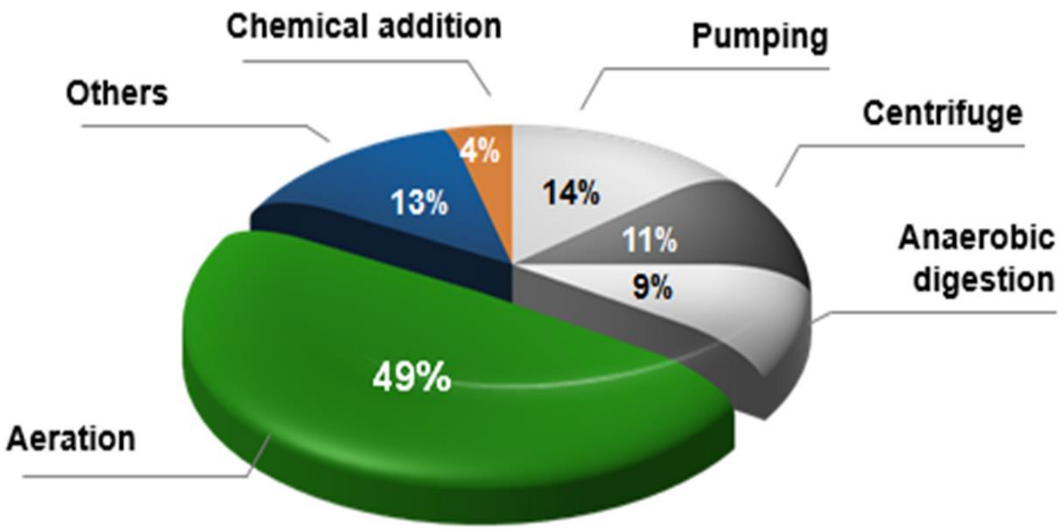

Fig 1.2. Estimated power usage for a typical 20MGD activated sludge facility performing wastewater treatment with nitrogen removal in the United States (adapted from MOP32, 2009).

1.2 Energy usage by different diffuser type

In a typical aerobic activated sludge treatment process, diffuser type, wastewater characteristics and oxygen transfer efficiency (OTE), are the potential parameters to monitor, because they have direct impact on the blower power and energy requirement for aerobic wastewater treatment. A thorough understanding of OTE for a process and how it fluctuates over time and seasons with the wastewater characteristics, will help aeration systems to be designed and optimized effectively to operate close to the actual oxygen demands of any process. This will also allow treatment plants to build flexibility into their aeration systems to address real-time oxygen demands to save on operational costs (USEPA, 2010).

Several types of aeration systems exist, but the two most popular ones are mechanical surface aeration and diffuse aeration systems. The former entrains air into the wastewater by agitation, thereby bringing wastewater in contact with the atmosphere, while the later transfers air or pure oxygen into the wastewater from diffusers installed at the bottom of the reactors. Diffuse aeration equipment used in the activated sludge process combine high volume, low air pressure blowers with air piping systems and diffusers installed at the bottom of the reactor to help diffuse the air supply into tiny bubbles (Stephenson et al. 2000; Van Der Roest et al, 2001; EPRI and WERF, 2013). This equipment is meant to provide the required dissolved oxygen for the biological processes and mixing to keep the solids suspended for effective treatment (Bolles, 2006; EPRI and WERF, 2013). Until recently, centrifugal and positive displacement types of blowers were most commonly used in wastewater aeration. In 2007, high-speed turbo energy-saving blowers were introduced in North America. These high-speed turbo blowers are more efficient and also have low maintenance costs compared to the centrifugal and positive displacement types, although they still have their limitations (EPRI and WERF, 2013).

Fine pore aeration diffusers have grown to become the most popular aeration technology employed in the activated sludge process, compared to coarse bubble diffusers and mechanical aeration systems. This is because fine pore diffusers produce bubbles which are less than 5mm in diameter with a higher surface to volume ratio and longer travel time, leading to higher oxygen transfer and aeration efficiency than other types of diffusers (Table. 1) (IWA, 2008). Fine pore diffusers are either made of polymeric membranes or ceramics.

Table 1. Different Diffuser Types and Their Standard Aeration Efficiency (SAE)/Power Consumed at Standard Conditions and Different SRT (Rosso and Stenstrom, 2010)

Aerator Type	SAE kg_{O_2} kWh^{-1}	Low SRT AE (@ 2 mg_{DO} l^{-1})	High SRT AE (@ 2 mg_{DO} l^{-1})
High-speed surface aerator	0.9–1.3	0.4-0.8	
Low-speed surface aerator	1.5–2.1	0.7–1.5	
Coarse-Bubble	0.6 –1.5	0.3-0.7	0.4-0.9
Turbines or jets (Fine-bubble)	1.2-1.8	0.4-0.6	0.6-0.8
Fine-Pore (Fine-bubble)	3.6–4.8	0.7–1.0	2.0–2.6

SAE- Standard Aeration Efficiency; **AE**- Aeration Efficiency; **SRT**-Sludge Retention Time

The different configurations of fine pore diffuser types in use today are membrane panels, ceramic discs, membrane discs and membrane tubes, with membrane panel diffusers having the advantage over the others in terms of higher OTE and an even distribution of fine air bubbles, though it is limited by high head loss and tear possibilities when over pressurized (USEPA, 2010). Although fine pore diffusers are known for their high efficiency per energy unit of energy used (Figure 1.3), they can be limited by fouling, scaling, aging, wastewater contaminants (i.e. surfactant), process and operating conditions when installed in biological reactors used for wastewater treatment. This decreases their OTE and aeration efficiency (Stenstrom et al, 1984; USEPA, 1989; Rosso and Stenstrom, 2006; IWA, 2008; USEPA, 2010; EPRI and WRF, 2013).

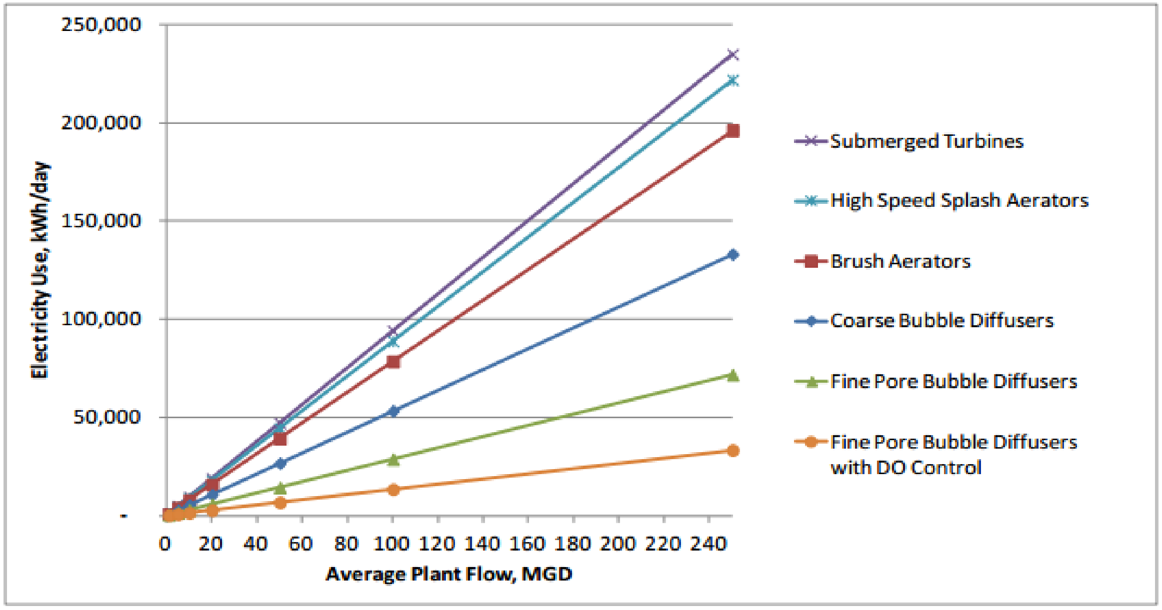

Fig 1.3. Electricity Use for a Variety of Aerator Types at a Range Of Plant Flow Rates (in kWh/day) (EPRI and WERF, 2013).

1.3 Aeration Efficiency Design Parameters and Oxygen Transfer Analysis Methods

A variety of parameters have been used to define oxygen transfer and its cost in aerobic biological processes. These are aeration efficiency (AE, $kgO_2\ kWh^{-1}$), volumetric mass transfer coefficient (K_La), alpha (α, %), alpha fouling (α F, %), oxygen transfer rate (OTR, kg d^{-1}), oxygen transfer efficiency (OTE, %), standard oxygen transfer rate (SOTR, kg O_2/hr), standard oxygen transfer efficiency (SOTE, %) for clean water and standard oxygen transfer efficiency under process condition (αSOTE, %). All these parameters are interrelated and used to quantify aeration efficiency, performance and their impact on plant energy consumption and cost (Redmon et al, 1983; Rosso and Stenstrom, 2006; ASCE, 2006).

Oxygen transfer analysis methods in wastewater aeration are best described by the two film theory proposed by Lewis and Whitman in 1924. The theory is based on adsorption or desorption of gas molecules estimated by diffusion across two stagnant films on the gas/liquid interface in a clean water test, and expressed as:

$$\frac{dC}{dt} = k_L a * (C_\infty^* - C) \tag{i}$$

where $k_L a$ = volumetric mass transfer coefficient

C_∞^* = saturated dissolved oxygen concentration

C = effective average DO concentration in the liquid phase

The $k_L a$ which is the overall mass transfer coefficient, is a combination of two factors, "k_L" which represents molecular diffusion resistance across the gas and liquid boundary, and "a" which is the surface area. In real wastewater treatment cases, $k_L a$ and the saturated dissolved oxygen concentration C_∞^* are influenced by different process parameters like temperature, bubble size, aeration tank depth, air flow rate, liquid depth, tank geometry, and water quality (Hwang and Stenstrom, 1985, Libra, 1993). Both $k_L a$ and C_∞^* are obtained from clean water testing (ASCE, 1993), which are also corrected to actual conditions during wastewater treatment. Oxygen transfer efficiency (OTE) has been evaluated through different methods like clean water testing, process water testing, material balance methods, and off-gas testing. However, off-gas testing has become more popular with proven benefits of accuracy over short testing periods. The off gas test analysis method as developed by Redmond et al, (1983) also has an advantage over the others in terms of error minimization from data estimation and conversion, as real time off gas data are obtain directly from the aeration systems. Oxygen uptake rate (OUR) which represents the microbial activity during aeration is also evaluated from the off gas test analysis based on the measured OTR, and it is expressed as the mass chemical oxygen demand (COD) per unit time (mgCOD/L/hr). However, no significant changes in oxygen uptake rate are observed during the activated sludge treatment process.

Off Gas Testing for OTE and Alpha Estimation

Off-gas testing is a real time measuring method for estimating air diffuser aeration efficiency defined by OTE and alpha (ASCE, 1997). The alpha Factor (α) is the ratio of standard oxygen transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency at standard condition of clean water (SOTE), and it accounts for wastewater contaminants (i.e. soap and detergent) which have an adverse effect on the oxygen transfer efficiency. The off-gas testing method evaluates OTE based on oxygen consumption estimation by comparing the oxygen content in the supplied air and captured off-gas. During off gas testing, off gas captured by a floating or stationed hood on the surface of an aeration tank or reactor (Figure 1.4) is treated to strip off CO_2 and water vapor, then the oxygen partial pressure is measured by an oxygen analyzer which is converted to percentage oxygen and analyzed as mole fractions of O_2.

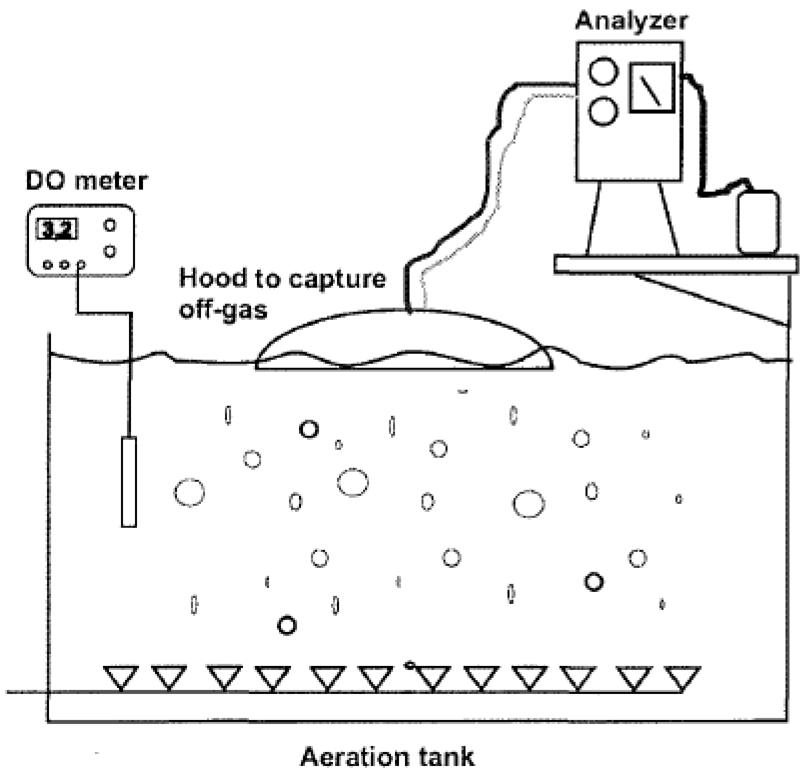

Fig 1.4 Schematic of off gas testing set up and measuring equipments (Shao-Yuan Leu, 2009)

The percentage of oxygen transferred to the process water is calculated as:

$$\text{OTE (\%)} = \frac{O_{2,in} - O_{2,out}}{O_{2,in}} \qquad \text{(ii)}$$

Where $O_{2,\%}$ = percentage oxygen in the supplied air.

The evaluated OTE is corrected to standard conditions of (20°C, 0 mg $_{DO}$/l, 1 atm, no salinity) to obtain a standardized OTE, or SOTE (%) (ASCE, 1984, 1991), and the alpha factor (α) is evaluated by taking the ratio of standardized OTE at process water conditions to clean water conditions. The oxygen transfer rate (OTR) is calculated by multiplying OTE by the measured airflow rate passing through the off gas hood.

From DO and OTR measurements, the gas mass transfer coefficient under wastewater process condition *($\alpha K_L a$)* can be evaluated as:

$$\alpha K_L a = \frac{OTR}{V*(C_{\infty}^{*}-C)} \quad \text{(iii)}$$

Where V = the volume of aeration tank,

α = ratio of gas transfer coefficient of process water to clean water conditions

Aeration Efficiency (AE): This the mass of oxygen transferred per unit of power input. It defines the amount of energy required to treat the wastewater, and is expressed as OTR divided by power input. Therefore, aeration efficiency is the most important parameter for describing the energy cost for air and oxygen transfer into wastewater, which involves shearing the liquid or releasing air through porous materials, to reach a defined concentration of dissolved oxygen.

Volumetric Mass Transfer Coefficient (K_La): It is a function of the aeration system and tank geometry. K_L represent mass transfer coefficient base on liquid film resistance, while the interfacial area of the gas bubble is represented by a.

Oxygen Transfer Rate (OTR): It's the mass of oxygen dissolved in the process water per unit time (i.e., kg/h or lb/h). At standard conditions (20^{o}C, 1atm, 0 DO, 0 salinity) it is designated as standard oxygen transfer rate (SOTR) and is the key process variable for aeration system design.

Oxygen Transfer Efficiency (OTE): It's the mass of oxygen transferred into the liquid from the mass of air or oxygen supplied, expressed as a percentage (%). OTE is the actual operating efficiency of an aeration system. At standard conditions of clean water (20^{o}C, 1atm, 0 DO, 0 salinity), it is designated as the standard oxygen transfer efficiency (SOTE) and at process conditions it is designated as (αSOTE).

Alpha Factor (α): It's the ratio of standard oxygen transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency at standard condition of clean water (SOTE). It is also referred to as the ratio of process water volumetric mass transfer coefficient to the clean water volumetric mass transfer coefficient. The alpha factor accounts for wastewater characteristics and other contaminants (i.e. soap and detergent) in the wastewater which have an adverse effect on oxygen transfer efficiency. The higher the alpha factor, the higher the OTR and at a low alpha factor the OTR will also be low. When a reduction in oxygen transfer efficiency is as a result of diffuser fouling or ageing, the fouling factor F is used to define that additional loss in OTE as a result of the fouled diffuser.

Alpha Fouling factor (α F): This is defined as the ratio of alpha after a period of time to alpha of a newly installed or cleaned diffuser. It therefore accounts for the decline in oxygen transfer efficiency over time, caused by diffusing fouling or ageing

1.4 Different approaches to energy savings.

The drive to achieving energy neutrality in advanced wastewater treatment plants (AWTP) has shifted focus to wastewater resource energy recovery to compensate for energy usage during wastewater treatment. Municipal nutrient removal plants have tried different wastewater resource energy recovery processes to attain this self-sufficient energy capabilities. Anaerobic digestion, co-digestion, carbon diversion, de-ammonification and aeration systems optimization are some of the popular wastewater resource energy recovery approaches being engaged. Nowak et al, 2011 reported that Strass (also called AIZ) and Wolfgangsee-Ischl treatment plants internally generated 6.3 % more electricity than they needed over a 3 years period after they added mesophilic anaerobic sludge digesters unit to their treatment plants. The energy was generated from biogas produced by feeding the anaerobic digesters with organic substrate and excess sludge, and supplied to the combined heat and power (CHP) units (Nowak et al, 2011). East Bay Municipal utility district, California currently generates 4.6 megawatts of electricity from co digestion and CHP turbines which they sell to the grid (Mark Ramirez, 2015).

Blue Plains (DC water), where this research was conducted, treats an average of 370 MGD of wastewater, and at peak flow treats 740 MGD of wastewater, consuming about 27 to 30 megawatt of electricity. The 6 two stage centrifugal compression blowers (4160 volts, 13000-HP) supplying air to 6 secondary aeration reactors with 4 passes each, consumes 14 % of the plant total energy consumption, while 20 % of the plant total energy consumption is consumed by another 6 two stage centrifugal compression blowers (4160 volts, 20000-HP), which supply air to 12 nitrification aeration reactors (Figure 1.5). In order to reduce the 34 % of plant total energy consumption by aeration in the secondary and nitrification stage, Blue Plains has proposed and are currently testing different wastewater resource recovery approaches (i.e., anaerobic ammonia oxidation (ANNAMOX) deammonification, anaerobic digestion and co-digestion). The annamox deammonification process is meant to reduce oxygen demand during nitrification by 67 %. The secondary and nitrification aeration reactors in Blue Plains use coarse and fine pore bubble diffusers respectively, for oxygen transfer. This research therefore is focused on aeration system optimization with an emphasis on fine pore diffusers and wastewater characteristics effects on oxygen transfer efficiency (OTE). Different treatment technologies are directly correlated to their effect on aeration performance.

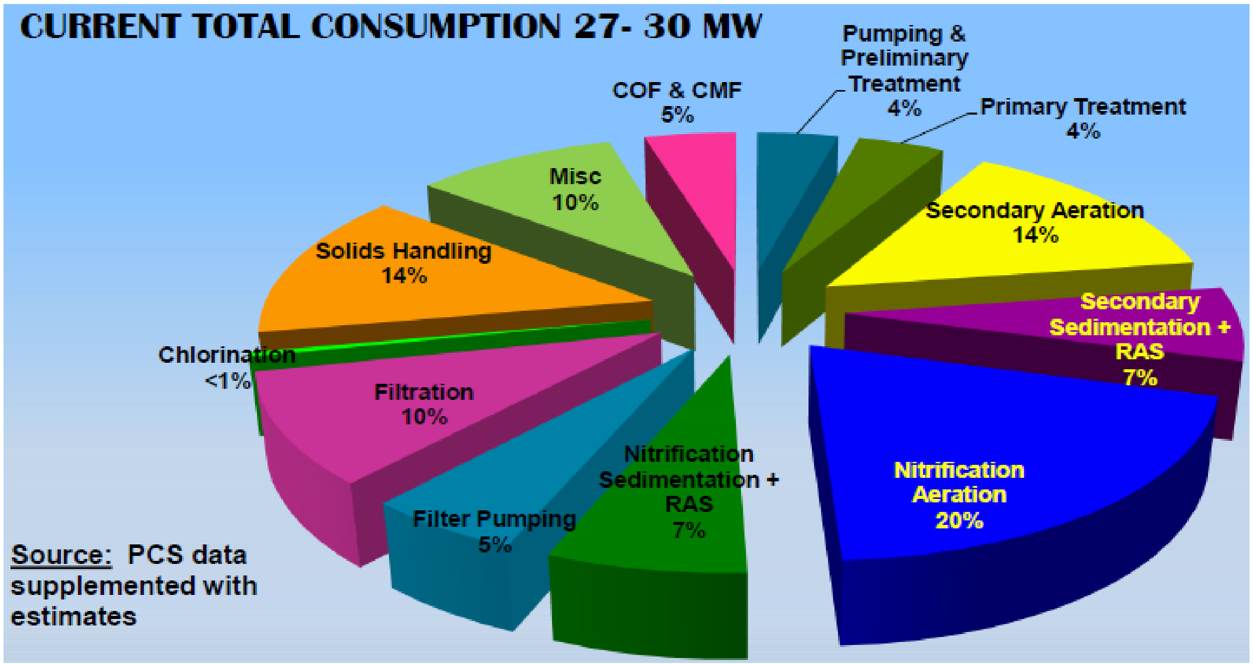

Fig 1.5 Estimated Blue Plain power usage for 370 MGD activated sludge facility performing wastewater treatment with nitrogen removal (DC Water Nutrient workshop, Mark Ramirez)

1.5 Fine Pore Diffuser Aeration efficiency and Application

The alpha factor is an important parameter used to measure diffuser OTE and it is dependent on diffuser fouling, sludge characteristics, process and operating parameter. Many wastewater treatment plants before the 1980's were designed with an alpha factor of 0.80, which was regarded as the universal alpha factor. Over the years, different studies have described the alpha factor as a function of different aeration techniques, and for the activated sludge process fine pore diffusers it's a function of process and operating conditions. The alpha factor is normally between 0.30 to 0.80 for plug flow aeration tanks and low and high sludge retention time (SRT) systems (Rosso et al, 2005; Rosso and Stenstrom, 2010). Fine-pore diffusers initial alpha factor decreases over time due to aging, fouling, inorganic scaling or changes as a result of wastewater quality, sludge characteristics and operating conditions (Rosso et al, 2008).

During aeration in the activated sludge process, mixed liquor or activated sludge requires dissolved oxygen (DO) for microbial activity, and the DO depends on OTE which is impacted by differential loading fractions, sludge characteristics or operational conditions (Germain and Stephenson et al, 2005). Mueller et al, (2002) stated that dissolved organics in wastewater have adverse effects on mass oxygen transfer, and the α-factor is used to account for these effects. Therefore an evaluation of the impact of sludge characteristics and operational condition on OTE, defined by oxygen transfer parameters (i.e. α-factor, αSOTE, K_La) is key towards understanding and optimizing aeration efficiency.

Fine Pore Diffuser Fouling Mechanism

Application of fine pore diffusers in wastewater treatments plants to save energy and operating cost has been limited by fouling, which has led to numerous research studies on causes of fine pore diffuser fouling, different fouling mitigation techniques and the impact on the alpha fouling factor. Le-Clech et al, (2006) did a comprehensive review on sludge characteristics, operational parameters and membrane materials as factors responsible for membrane fouling, and concluded that no known single parameter can predict or model fouling of fine pore diffusers due to changes in biomass characteristics from plant to plant. But an understanding of foulants and their interaction with diffuser polymeric material may provide a new direction for their cleaning and for mitigation strategies. Diffuser fouling, also known as organic fouling, can be defined as the biofilm coating or plugging of diffuser surface and pores by a combination of colloids, microbial suspended solids, and other smaller molecules. Scaling, known as inorganic fouling, is the precipitation of inorganic salts on diffuser surfaces. Both types of fouling, which normally occur together over time and at increased influent loadings, causes a reduction in the diffuser membrane pore permeability due to blocked pores, leading to an increased pressure drop across the membrane diffuser and uneven large bubble production, which in turn affects the oxygen transfer efficiency and aeration performance (Boyle and Redmon, 1983; Kim and Boyle, 1993). Fine pore diffuser fouling has been grouped into three category of bio, organic and inorganic fouling by a previous study (Meng et al. 2009). They defined biofouling as the deposition, growth and metabolism of bacteria cells or flocs on diffuser membrane surfaces. Other studies reported how selected bacteria with higher hydrophobicity than suspended sludge selectively adhere and grow on diffuser surfaces, making their fouling non-removable (Jinhua et al. 2006; Miura et al. 2007).

Organic fouling has been defined as the deposition of biopolymers (i.e., proteins and polysaccharides) on membrane surfaces. The deposited biopolymers were found to be composed of three different layers; a loosely bound cake layer similar to the sludge floc which causes removable fouling, an intermediate layer with high polysaccharides, bacteria aggregates and exocellular microbial products which causes less removable fouling, and finally a lower layer with a high concentration of exocellular microbial products and proteins which also causes non-removable fouling (Metzger et al. 2007).

Inorganic fouling which is the last category of fouling is defined as the chemical and biological precipitation of numerous cations and anions. Biopolymers contain ionisable groups (i.e. COO^-, CO_3^{2-} SO_4^{2-}, PO_4^{3-}, and OH^-) which are easily captured by metal ions (i.e. Ca, Fe, and Mg) present in wastewater. They form dense cake layers with deposited bacterial cells and biopolymers on membrane surfaces to cause non-removable fouling (Lyko et al. 2007; Costa et al. 2006; Wang et al. 2008b; You HS et al. 2005; You HS et al. 2006).

Removable fouling as described by Meng et al. 2009 is caused by loosely bound foulants on membrane surfaces and can be removed by physical cleaning like backwashing. Less easily removable fouling is caused by fine pore clogging and strongly attached foulants to membrane surfaces due to their affinity for the membrane, and it can only be removed through chemical cleaning.

Fouling quantification

Previous studies have reported dynamic wet pressure (DWP) as the major indicator of fouling in fine pore diffusers by quantifying the differential pressure drop across the membrane over time of operation (USEPA, 1989; Kim and Boyle, 1993; Rosso et al. 2008). DWP is defined as the sum of friction to air flow through diffuser pores and the surface tension of the fluid being aerated.

Also, air blowers designed for maximum air flow have a discharge pressure requirement that is meant to only accommodate new DWP, and water head hydrostatic pressure which is always constant through the aeration process. Therefore as the DWP of the fine pore diffusers increases due to fouling, there is a demand on the blower discharge pressure, which results in most aeration systems having blowers with insufficient discharge pressure, which produce uneven bubbles. Because they primarily discharge air through diffuser pores with a lower head loss, this limits OTE and DO which impacts aeration performance and may result in odor formation and create sludge mixing/bulking problems (Palm et al. 1980; Rosso et al. 2008). A previous study (Rosso et al. 2008) defined a pressure factor P (ratio of used diffuser DWP to new diffuser DWP) to represent the pressure burden to the blower caused by fouling or ageing.

In summary, fine pore diffuser fouling as described by different researchers is primarily caused by wastewater characteristics which form different kinds of foulants that are deposited on the diffuser surface or plug diffuser slits and pores. Some researchers classified these foulants as organic and inorganic, while others classified them as organic, inorganic and biofoulants which cause reversible or less reversible fouling (Le-Clech et al. 2006; Meng et al. 2009; Boyle and Redmond, 1983; Kim and Boyle, 1993). Diffuser fouling as reported by the different researchers leads to an increase in DWP of the diffusers and a decrease in their OTE (Cheng et al. 2000;

Hansen et al. 2004; Wagner and Von Hoessle, 2004; Rosso et al. 2007). There is insufficient information on the impact of diffuser fouling (reversible or irreversible) and ageing, with respect to diffusers increasing DWP and decreasing OTE. This has made it difficult for design engineers to select an approach to effectively prevent diffusers increasing DWP and decreasing OTE that contributes to increased aeration costs over time of operation. Further investigation is needed to fill this void and is addressed in Chapter 1 of this study which focuses on long term diffuser fouling monitoring and the application of a fouling mitigation techniques aimed at removing the different type of fouling (reversible and irreversible), with respect to their impact in preventing DWP increase and OTE decreases that contributes to aeration cost. Process variables were also measured during this study to ascertain their impact on fouling.

Fouling Mitigation Techniques

Past studies have established that fine pore diffusers experience fouling and consequently experience increases in DWP and OTE decline within the first 12 to 24 months of installation or after cleaning, depending on the materials used, their interaction with wastewater characteristics, plant operating conditions and time of operation. In order to mitigate fine pore diffuser fouling and its effect on plant energy and operational costs, routine cleaning is a necessity (Stenstrom et al, 1984; USEPA, 1989; Rosso and Stenstrom, 2006; IWA, 2008; EPRI and WRF, 2013).

Over the years, both physical and chemical cleaning methods have been applied as fouling mitigation technique to fine pore diffusers. Some of the physical cleaning methods applied in previous studies include gas or air sparging, membrane relaxation, pressure and backwashing or flushing (Schiewer et al. 2005; Schiewer et al. 2006; Le-Clech et al. 2006; Meng et al. 2009; USEPA, 2010).

Most manufacturers have different proposals for fine pore diffuser fouling mitigation by chemical cleaning, which normally differ by the choice of chemical compounds used and the concentration and application frequency. In general, the choice of cleaning chemicals depends on wastewater feed characteristics. Acidic cleaning chemicals are most suitable for removing precipitated salts while alkaline cleaning chemicals are suitable for absorbed organic removal (Van der Bruggen et al, 2003). The chemical cleaning method deals with less removable fouling according to some studies, although until now, no known protocols for the use chemical cleaning agents has been published (USEPA, 2010, Rosso et al, 2012; Lee and Kim, 2013).

In summary, most of the physical cleaning method applied for fouling mitigation are limited to membrane filters and not diffusers. There is also insufficient information on the effectiveness of the different types of fouling mitigation techniques with respect to curbing DWP escalation and OTE decrease. Chapter 1 of this research tries to bridge that gap by studying the effectiveness of both physical and chemical treatment methods in fouling mitigation with respect to preventing DWP increase and OTE decrease. This provides useful information to design engineers on the effectiveness of treatments method.

1.6. Wastewater characteristics impact and the indirect influence of OUR on alpha

The importance of oxygen transfer for microbial activity and metabolism cannot be over emphasized, because the OTR of an aerobic process is strongly influenced by hydrodynamic conditions (OUR, K_La) in the aerobic process. However, the hydrodynamic conditions depend on the operating conditions (SRT, air flow, CEPT etc.), wastewater physicochemical characteristics (readily biodegradable COD and slowly biodegradable COD) and the presence of oxygen-consuming cells (bacteria) and oxygen transfer reducing contaminants (surfactants) in the aerobic system. The impact of wastewater characteristics and operating conditions on the hydrodynamic parameters is measured by the alpha. Therefore an understanding of alpha and OUR correlations with the wastewater characteristics and operational conditions is of great importance in the design and optimization of any aerobic activated sludge system. Wastewater characteristics and operational parameters (i.e., organic load, surfactant, SRT, MLSS and MLVSS) influence the oxygen transfer rate (OTR) and alpha factor based on the interfacial mass transfer at the gas to liquid interface and impurities present in the wastewater (Rosso et al. 2005). The soluble COD and its readily biodegradable fraction (rbCOD) reduce the alpha factor for fine-pore diffusers by depressing the SOTE under process conditions (Hwang and Stenstrom, 1979). The lowest alpha factors were also found at high organic loadings and high surfactant concentrations (Leu et al. 2009). In addition, a surfactant (sodium lauryl sulfate) concentration of 15 mg/L in wastewater decreased the oxygen mass transfer coefficient by 50% of the value in the clean water test (Eckenfelder et al. 1961). High alpha values are generally observed for longer SRT conditions and low food to microorganism (F/M) ratios (Rosso et al., 2005). An indirect effect of OUR on alpha was reported by Riber and Stensel (1985). They found that an increasing bulk DO concentration resulted in a decrease in interfacial OTR from gas to liquid phase, suggesting a potential effect of low OUR on the OTR. Other studies have also reported an OTE increase with an increasing OUR, suggesting the possibility of an alternate direct oxygen transfer pathway by which microbes bypass the bulk DO and use oxygen from the gaseous phase (Bartholomew et al, 1950; Albertson and Digregorio, 1975; Eckenfelder et al, 1979). Adsorption of microbial cells to gas bubbles in the bulk liquid potentially increases the interfacial transfer rate (OUR) between the microbial cells and gas bubbles and consequently increases the oxygen transfer rate (Tsao et al, 1968). An increase in the COD fraction of soluble microbial product (SMP) was found to decrease alpha and K_La, due to surfactants in sludge contributing to the COD. However, the investigation by Stenstroms (1989) on the impact of biomass MLVSS on OTE in a full scale study from 6 different conventional wastewater treatment plants, reported a good correlation between MLVSS and the alpha factor, where the average alpha values at the aeration basin inlet were observed to increase with increasing MLVSS concentration.

In another study of a plug flow aeration system, soluble COD (SCOD) was found to be a principal factor affecting OTE and alpha (Muellar et al. 2000). Leu et al. (2009) also in their study during a 24 hours full scale plant transient test, reported the lowest alpha factor when the COD load was highest, and attributed it to surfactants that are present in the influent which contributes to the COD load.

Gas transfer processes in liquids are also inhibited by surface active contaminants dissolved in the liquid medium. They accumulate at the gas-liquid interface and reduce mass transfer rates. Therefore investigating their effect on OTE and alpha is important in designing and optimizing aeration systems to save energy cost (Rosso, 2005). Wagner et al. (2006) reported in their study that OTE and the alpha factor were a function of sludge type, surfactant concentration and surfactant type. They recorded stronger inhibition of OTE and alpha with nonionic surfactants compared to anionic surfactants, although they limited their test to synthetic (manufactured) surfactant solutions and a clean water matrix only without testing in process water. Hebrard et al. (2014) also reported an enhanced inhibition of mass transfer at the gas-liquid interface by increasing the dosage of synthetic surfactant, which formed a barrier at the gas-liquid interface and prevented mass transfer.

In summary, different sludge characteristics (COD, SCOD, MLSS, MLVSS and surfactants) have been reported to impact alpha. Some of these studies contradicted each other with conflicting reports of decreasing alpha with increasing MLSS and SCOD (Duran et al. 2015; Muellar et al. 2000; Leu et al. 2009), while other studies reported either increasing alpha or no change respectively (Stenstrom et al. 1989; Stephenson et al. 2007). There is also insufficient information on the impact of low to average MLSS concentration (range expected in a typical conventional wastewater treatment plant) on alpha, as most of the results reported by previous studies were at high MLSS concentration attainable only in MBRs. Chapter 3 of this study addressed this issue by running different batch experiments to determine the impact of wastewater characteristics and concentrations on alpha. Chapter 3 reports on the impact of different wastewater treatment stages in a full-scale plant, organic load (synthetic rbCOD, sbCOD and surfactant) and MLSS concentrations on alpha and OUR.

1.7 Activated sludge different process treatment technology and their impact on OTE and alpha

Presently, the Blue Plains Advanced Wastewater Treatment Plant (AWTP), Washington DC where this research study was conducted treats about 1,458,333 m^3/d (384 million gallons per day) wastewater. The raw wastewater is first treated by passing it through screens and an aerated grit removal chamber, before it flows into the chemically enhanced primary clarifier. The effluent from the chemically enhanced primary treatment (CEPT) clarifier, is then treated for carbon removal through a biological step-feed high rate activated sludge (HRAS) treatment process. The high-rate activated sludge (HRAS) system treats high load sewage under a short period of continuous aeration according to a past study (Greeley and Dixon, 1943). The HRAS systems secondary reactors at Blue Plains are operated at short SRTs of 1.5-2 days, a residual dissolved oxygen (DO) of 0.5-1 mg/L and a targeted average total suspended solid (TSS) of 2500 mg/L. The effluent from the HRAS clarifier flows to the nitrification/denitrification reactor operating at an SRT of 15 days, for biological nitrogen removal (BNR). Thereafter, an enhanced nitrogen removal (ENR) process is used to remove any residual ammonia still present before it goes to the nitrification/denitrification clarifier. The effluent from the clarifiers is passed through the multimedia filtration and disinfection phases before they are discharged to the Potomac River (Figure 1.6).

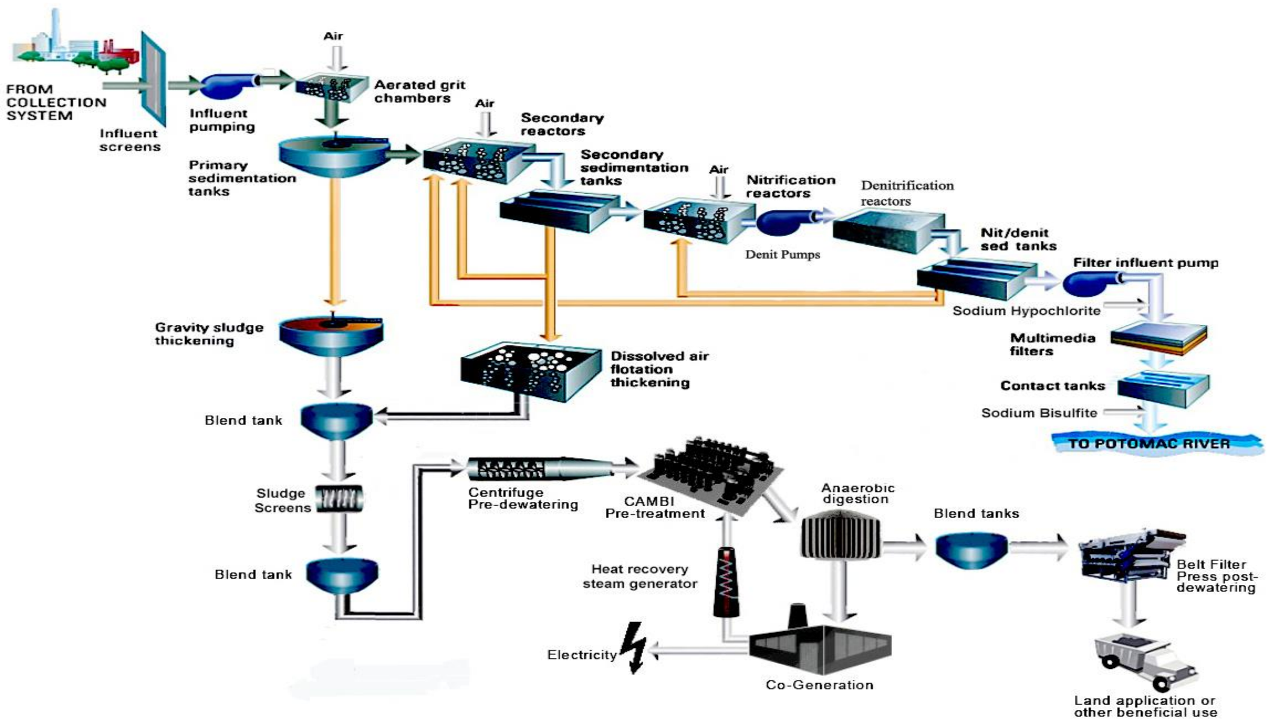

Fig 1.6. Schematics of Blue Plains Advanced Wastewater Treatment Plant Processes. (DC Water Nutrient workshop, Mark Ramirez)

The HRAS secondary reactors are separated into west (bioaugmented) and east (non bioaugmented) secondary reactors. About 50 % of waste activated sludge (WAS) from the biological nutrient removal (BNR) or nitrification/denitrification stage is recycled to the west reactor for bioaugmentation as captured by the golden lines in Figure 1.6 and Figure 1.7, while the east (non bioaugmented) reactor operates normally without receiving any WAS recycling. Both secondary HRAS reactors (east and west) have an anaerobic selector phase of about 45 mins HRT and zero dissolved oxygen at the beginning of each reactor to enhance sorption, carbon capture and phosphorus removal. The aerobic phase of the HRAS reactors is installed with coarse bubble diffusers which operates in parallel with a residual DO of 0.5 to 1 mg/L,

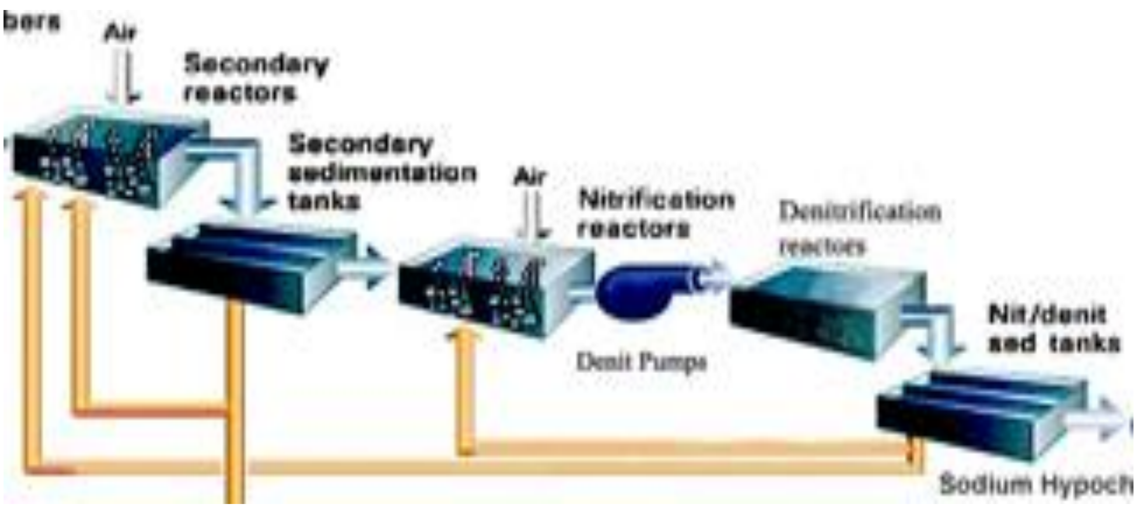

Fig 1.7. Schematics of Blue Plains Advanced Wastewater Treatment Plant High Rate Activated Sludge and Nitrogen Removal Processes. (DC Water Nutrient workshop, Mark Ramirez)

The activated sludge different process treatment technology employed at DC Water (HRAS, Bioaugmentation and Anaerobic selector) for biosorption and biodegradation of organic contaminants during wastewater treatments were evaluated in chapter 4 of this study, and correlated with their impact on surfactant removal, OTE and alpha. Another activated sludge treatment process evaluated in this chapter was the contactor-stabilization treatment process with high biosorption capacity of organic carbon as reported in a past study (Rahman et al., 2015)

Contactor Stabilization Treatment Process.

The contact-stabilization (CS) treatment process is designed to enhance the biosorption capacity of the activated sludge in order to improve the overall carbon capture from wastewater (Rahman et al., 2015). The treatment process consists of a stabilizer reactor and a contactor reactor where return activated sludge (RAS) from the secondary clarifiers is aerated in the stabilizer reactor under high aerobic condition.

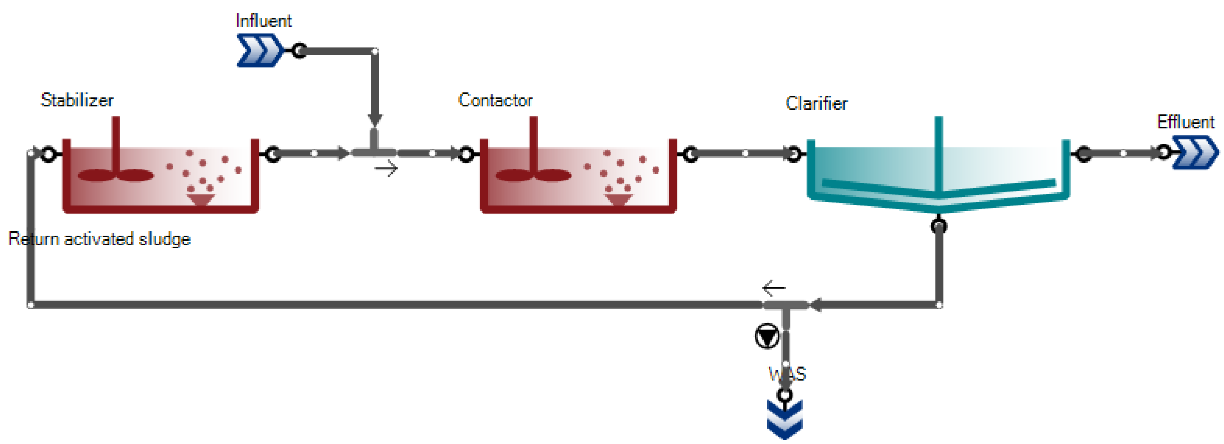

Fig 1.8. Typical Contact Stabilization (CS) process flow reactor configuration

The goal for the stabilizer is to oxidize adsorbed particulate material, colloidal COD and soluble intracellular COD from carbon-rich RAS, causing famine or starvation of the microbes in the RAS, and allowing for an increased organic sorption/uptake capacity when recycled back to the contactor (Meerburg et al., 2015; Rahman et al., 2015; Vasquez Sarria et al., 2011). The contactor reactor receives raw influent feed and the starved aerated RAS from the stabilizer under anaerobic or low DO conditions. An enhanced biosorption of organic carbon is achieved in the contactor reactor, where the starved RAS microbes with an increased surface area comes in contact with the organic carbon rich influent feed. Microorganisms leaving the contactor and going to the clarifier are partly harvested as waste activated sludge (WAS) and sent to an anaerobic digester for biogas production. The remaining fraction of settled biomass is sent back to the stabilizer. However, the CS technology energy saving potentials of good effluent quality, sludge bulking resistance and sufficient biomass inventory (Kraus, 1955) has not been effectively utilized with respect to the HRAS system for organic contaminant removal through enhanced biosorption during wastewater treatment. The impact of this study on aeration efficiency has also not been adequately reported, which is one of the objectives of this study.

Bioaugmentation Treatment Process

Bioaugmentation, which is the addition of a special bacteria or return of nitrifying sludge to the secondary or HRAS treatment stage (Figure 1.9), enhances the removal of contaminants of concern and recalcitrant substrates (i.e. surfactant) through biosorption and biodegradation, which can potentially improve aeration efficiency (Babcock et al., 1993; Parker and Wanner, 2007).

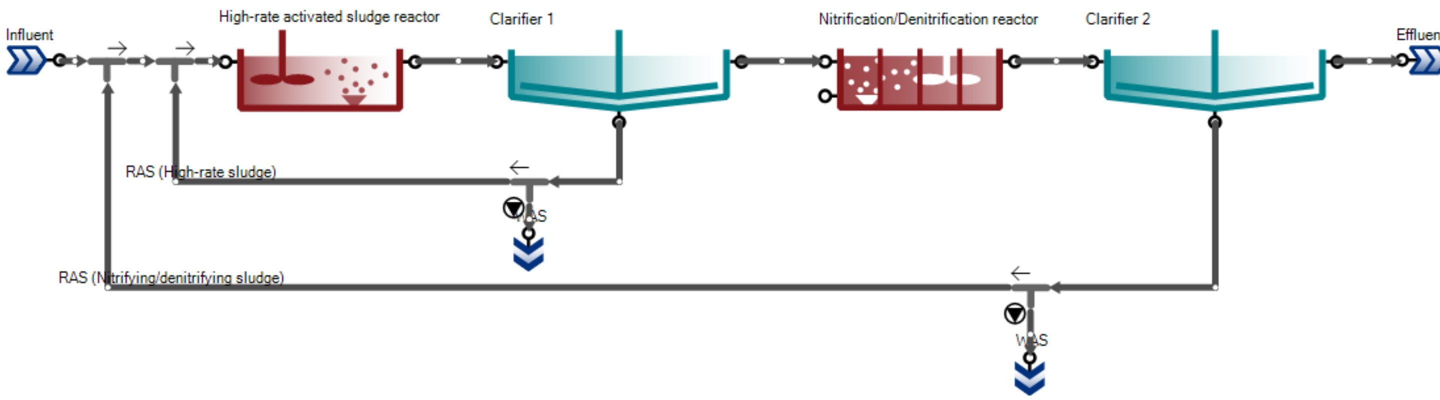

Fig 1.9. Typical bioaugmented HRAS process flow reactor configuration

Past studies have reported enhanced surfactant and colloids sorption capacity and degradation through the bioaugmentation treatment technology, which makes it a potential tool and cost-effective method for degradation of recalcitrant compounds (Parker and Wanner, 2007; Limbergen et al., 1998). Grubbs (1974, 1983) reported a decrease in sludge and foam production in the aeration process, and the suppression of filamentous organisms in the activated sludge when using bioaugmentation, which also improved the effluent quality. Until now, little information on its impact on aeration efficiency has been provided. A better understanding of the role of bioaugmentation in improving aeration efficiency will help engineers to design and optimize aeration systems to save energy and operating costs.

Anaerobic Selector Treatment Process

The process of bioselection technology is now used for many activated sludge treatment processes. At Blue Plains the secondary HRAS reactors has an anaerobic selector phase of about 45 mins HRT and zero dissolved oxygen at the beginning of each reactor before the aerobic phase (Figure 1.10) to enhance sorption, carbon capture and phosphorus removal.

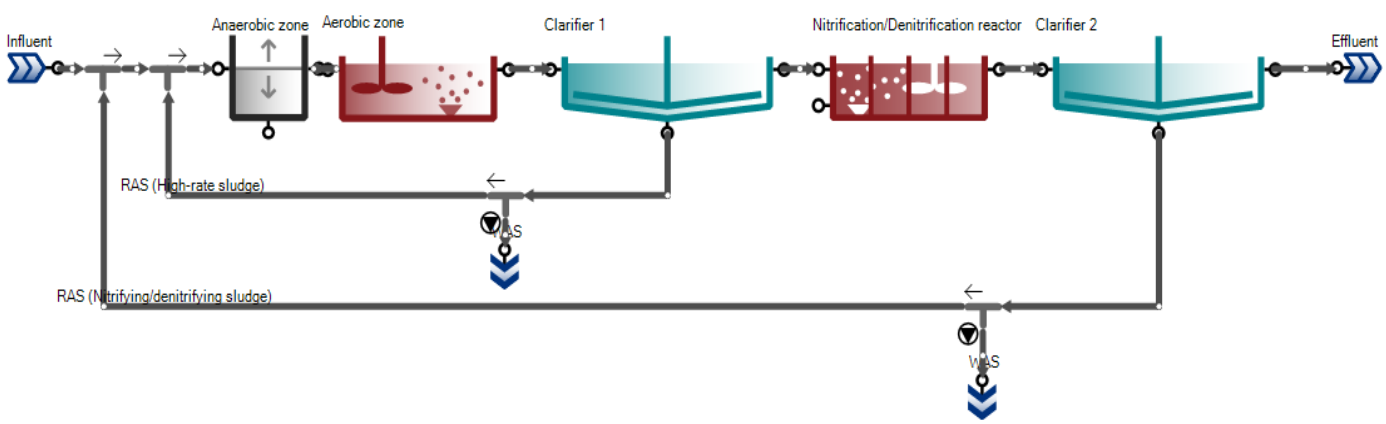

Fig 1.10. Typical Anaerobic selector phase (An-S) HRAS process flow reactor configuration

However, there are little information on the impact of hydraulic retention time of individual bioselector (aerobic, anoxic or anaerobic) basins. While bioselection is successful in reducing the sludge volume index (SVI) in nearly every case, the level of performance can vary widely. According to some studies (Albertson, 2005; Ali Akbar Azimi and Mirzaman Zamanzadeh, 2006), an anaerobic selector reduces soluble COD and enhances sorption capacity during wastewater treatment. Albertson et al., 2005 in their study reported that an anaerobic selector reduced soluble COD to the range of the final effluent in 12 to 30 minutes of the retention time based on raw or settled effluent flow.

Presently, there is insufficient information on the impact of specific technologies (HRAS, CS, Bioaugmentation and Anaerobic selector) with respect to their impact on aeration efficiency (i.e., OTE and alpha). Chapter 4 of this study focusses on bridging this gap.

1.8 Research Objectives

The energy requirement for aeration in the activated sludge process, which also includes nitrogen removal, is a major operating expense for utilities and it limits the ability of most water and wastewater reclamation facilities to achieve energy neutrality. Aeration has therefore become one of the most energy and capital intensive aspects of wastewater treatment. The objectives of this research are geared towards resolving the major critical challenges of energy consumption through aeration in the carbon and nutrient removal stage of the activated sludge treatment process. The study provides an insight into key activated sludge processes and operational parameters contributing to aeration cost (Figure 1.5). The impact of new and optimized activated sludge treatment technologies on oxygen transfer contaminants directly correlates to their impact on aeration efficiency. The goal is to provide adequate information to prevent the overdesign and consequent energy wastage in the cost of the activated sludge wastewater treatment process in order to achieve an energy neutral (positive) wastewater treatment process.

The specific objectives are listed as follows:

Objective 1: Evaluation of diffuser fouling dynamics, its impact on OTE and DWP, and the development a fouling mitigation techniques that can prevent or reduce fouling.

Objective 2: Determine the impact of different wastewater characteristics on aeration efficiency design parameters, OTE and alpha.

Objective 3: Evaluate three different optimized activated sludge treatment technologies to reduce surfactants through enhanced biosorption and biodegradation.

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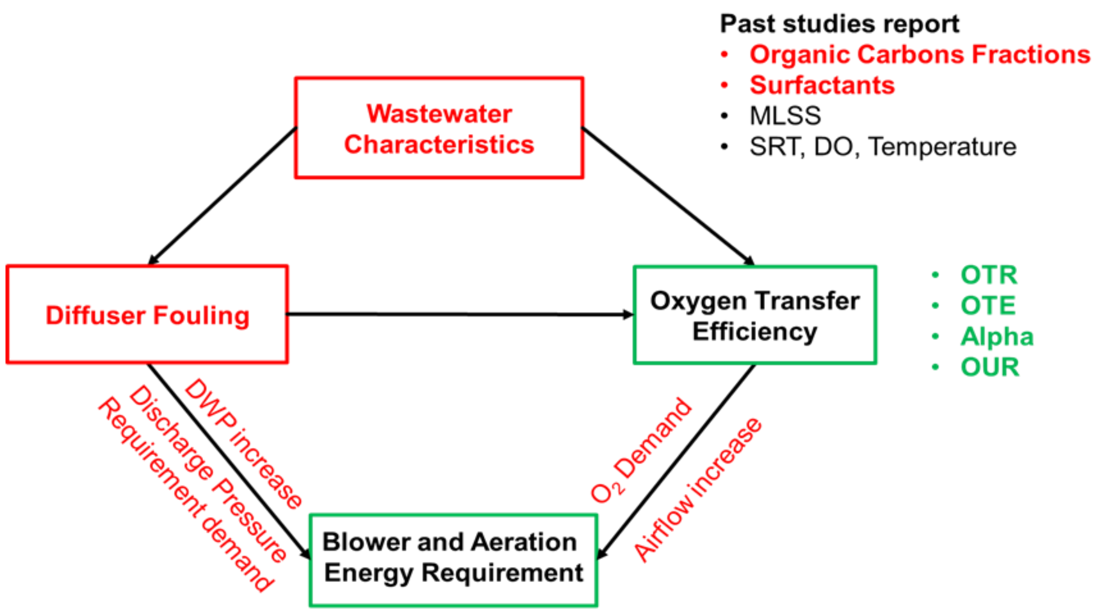

Fig 1.11: Diagram showing key components contributing to aeration cost (red) and design parameters (green) to improve on to minimize energy consumption and aeration cost.

1.9 ANNOTATED DISSERTATION OUTLINE

Chapter 1: ***Introduction:*** This chapter provides the research study background and defined the research focus by stating the four major research objectives. A graphical abstract for the experimental work (Figure 1.6) and brief summary of the material presented in this dissertation as included in the annotated dissertation outline are also provided in this chapter.

Chapter 2: ***Mechanical Cleaning/Treatment Method for Mitigating Membrane Diffuser Fouling and Improving Aeration Efficiency:*** This chapter investigated fine pore diffuser fouling dynamics, its defouling efficiency through a mechanical/physical cleaning technique (reverse flexing, RF). The study investigated three polyurethane fine-pore membrane panel (MP) diffusers installed in a test reactor column, which was continuously fed with mixed liquor from the plant's secondary stage. The use of a mechanical cleaning method (reverse flexing, RF, i.e. the periodic release of the air-side pressure to collapse the membranes) on the fine pore membrane diffusers, and its impact on oxygen transfer efficiency and alpha were measured and evaluated in a large pilot scale reactor. Air flow rate to the MP was regulated at 6SCFM (corresponding to 0.56 SCFM/ft2diff) to transfer oxygen to the reactor. Two of the installed diffusers (MP2 and MP4) were reverse flexed (RF) while the other diffuser (MP1) served as a control without reverse flexing. Dynamic wet pressure (DWP) measurement and off gas testing of all three diffusers were used to determine the diffuser fouling and mitigation impact on aeration efficiency design parameters, alongside other process parameters/operating conditions measurements.

Research objective 1 was addressed in this chapter which has also been published in Water Science and Technology.

Odize, V.O., Novak, J., Rosso, D., De Clippeleir, H., Al-Omari, A., Garrido, M., Smeraldi, J., Murthy, S. (2017). Reverse Flexing as a Physical/Mechanical Treatment to Mitigate Fouling of Fine Bubble Diffusers, Water Science & Technology 76 (7), 1595 – 1602.

Chapter 3: ***Impact of Wastewater Matrix Characteristics on Oxygen Transfer Efficiency:*** This study investigated the influence of a full-scale plant different wastewater characteristics/composition matrices (treatment phases) on oxygen transfer efficiency (OTE) and alpha. A strong relationship between the wastewater matrices was established, and as expected increased alphas were observed for the cleanest wastewater matrices (i.e., with increased treated effluent quality). A batch test conducted to characterize the mechanistic impact of the wastewater contaminants present in the different wastewater matrices identified that the major contaminants influencing OTE and alpha were surfactants and particulate/colloidal material. Measurements of k_La also showed the same results, thereby providing new tools as well as critical factors impacting OTE and alpha.

Research objective 2 was addressed in this chapter which was presented in WEFTEC 2016 conference (Appendix A). This chapter manuscript is also ready to be submitted for publication to the journal, Water Research.

Impact of Organic Carbon fractions and Surfactants on Oxygen Transfer Efficiency
Victory Odize, Haydee De Clippeleir, John T. Novak, Ahmed Al Omari, Arifur Rahman, Diego Rosso and Sudhir N. Murthy. Proceedings of the WEFTEC 2016, New Orleans, LO.

Victory Odize, John T. Novak, Ahmed Al Omari, Arifur Rahman, Diego Rosso, Sudhir N. Murthy and Haydee De Clippeleir, (2017) Impact of Wastewater Matrices and Characteristics on Alpha and Oxygen Transfer Efficiency in an Activated Sludge (Ready for submission for Water Research)

Chapter 4: ***Evaluation of strategies or technologies enhancing surfactant biosorption and biodegradation to improve OTE and Alpha in the High rate activated sludge system:*** This study was designed to evaluate 4 different wastewater secondary treatment strategies/technologies that enhance surfactant removal through enhanced sorption and biodegradation and also determine their effect on oxygen transfer and alpha. Different pilot and batch scale studies were conducted to compare and correlate surfactant removal efficiency and alpha of conventional high-rate activated sludge (HRAS), optimized HRAS with contactor-stabilization technology (HRAS-CS), optimized HRAS bioaugmented (Bioaug) with nitrification sludge (Nit S) at different ratio and optimized bioaugmented HRAS with an anaerobic selector phase technology (An-S) reactor system configuration**.**

Research objectives 3 and 4 were addressed in this chapter and a manuscript from this chapter is also ready to be submitted for publication in the journal, Water Research.

Victory Odize, John T. Novak, Ahmed Al Omari, Diego Rosso, Sudhir N. Murthy and Haydee De Clippeleir, (2017) Evaluation of strategies or technologies enhancing surfactant biosorption and biodegradation to improve OTE and Alpha in the High rate activated sludge system (Ready for submission for Water Research Journal)

Chapter 5: ***Concluding remarks and engineering significance.*** The concluding chapter discusses specific significant contributions of this research to the wastewater engineering field and proposes future research recommendations. This chapter aim to summarize the impacts of each study and its significance in improving aeration efficiency when applied to plant full scale operation, with the aim of saving aeration and blower energy requirements and cost.

1.10 EXPERIMENTAL OUTLINE

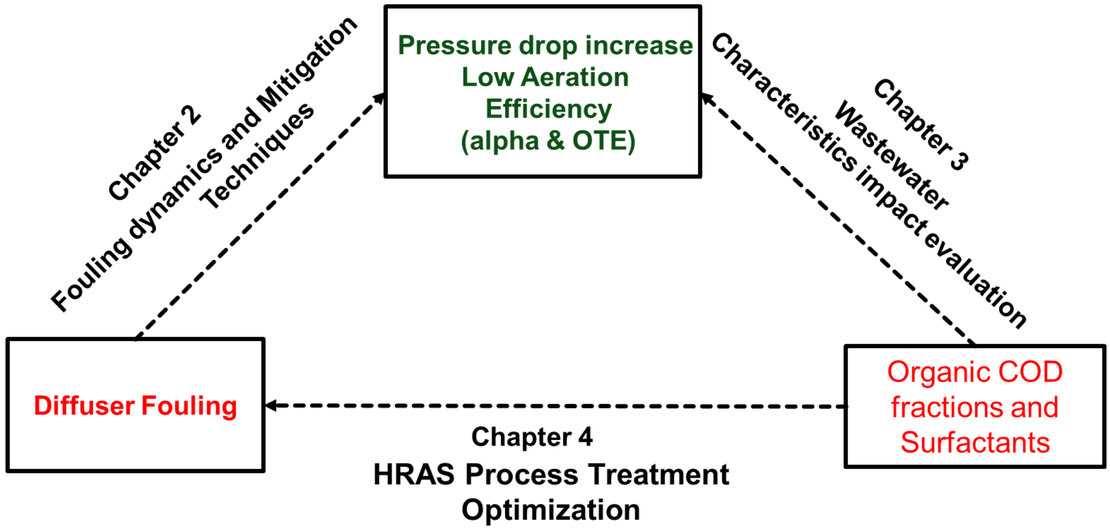

Fig 1.7: Dissertation experimental outline highlighting aeration efficiency challenges and studies to address them

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CHAPTER TWO

REVERSE FLEXING AS A PHYSICAL/MECHANICAL TREATMENT TO MITIGATE FOULING OF FINE BUBBLE DIFFUSERS

2.1 AUTHORS

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2.2 ABSTRACT

Achieving energy neutrality has shifted focus towards aeration system optimization, due to the high energy consumption of aeration processes in modern advanced wastewater treatment plants. A study on fine bubble diffuser fouling and mitigation, quantified by dynamic wet pressure (DWP), oxygen transfer efficiency and alpha was carried out in Blue Plains, Washington, DC. Four polyurethane fine bubble diffusers were installed in a pilot reactor column fed with high rate activated sludge from a full scale system. A mechanical cleaning method, reverse flexing (RF), was used to treat two diffusers (RF1, RF2), while two diffusers were kept as a control (i.e., no reverse flexing). There was a 45% increase in DWP of the control diffuser after 17 months of operation, an indication of fouling. RF treated diffusers (RF1 and RF2) did not show significant

increase in DWP, and in comparison to the control diffuser prevented about 35% increase in DWP. Hence, reverse flexing potentially saves blower energy, by reducing the pressure burden on the air blower which increases blower energy requirement. However, no significant impact of the RF treatment in preventing a decrease in alpha-fouling (αF) of the fine pore diffusers, over time in operation was observed.

2.3 KEYWORDS

Alpha, dynamic wet pressure, fouling, reverse flexing

2.4 INTRODUCTION

Energy consumption of aeration processes in modern advanced wastewater treatment plants comprises 45 to 75% of the plant total energy requirement, making aeration the most energy intensive aspect of wastewater treatment (Houck & Boon 1981; Rosso & Stenstrom 2005). Fine bubble diffusers compared to coarse bubble and other mechanical aeration diffusers, have grown to become the most popular aeration technology employed in municipal activated sludge processes. This is because fine bubble diffusers produce bubbles which are less than 5 mm in diameter and therefore have a higher surface to volume ratio and longer travel time. This leads to higher oxygen transfer and standard aeration efficiency (SAE) than other diffuser types. An SAE of 0.7–1.0 kgO2 kWh^{-1} for fine bubble diffusers compared to 0.3–0.7 for coarse bubble diffusers at a low dissolved oxygen (DO) of 2 mg/L) have been previously reported (IWA 2008). However, application of fine pore diffusers in wastewater treatments plants to save energy and operating cost has been limited by fouling. This has led to numerous research studies on causes of fine bubble diffuser fouling, different fouling mitigation techniques, their impact on oxygen transfer efficiency (OTE) and alpha fouling factor. Alpha fouling factor is a parameter that indicates the fouling impact on OTE. Diffuser fouling as reported by the different researchers (Boyle & Redmon 1983; Kim & Boyle 1993; Rosso et al. 2007) normally occurs over time and at an increased influent loading. It also increases the dynamic wet pressure (DWP) across the membrane diffuser, with an adverse blower effect and large bubble production, which results in a decrease in OTE (%) and alpha factor (Cheng et al. 2000; Rosso et al. 2007; Wagner & von Hoessle 2003). Other studies have established that fine bubble diffusers experiences fouling, and consequent DWP increase and OTE decrease within the first 12 to 24 months of installation or after cleaning (USEPA 1989; Rosso & Stenstrom 2006a, 2006b; IWA 2008; EPRI 2013). According to these studies, the fouling process is dependent on the materials they are made of, their interaction with wastewater characteristics, plant operating conditions and time in operation. In order to mitigate fine bubbles diffuser fouling and its effect on plant energy and operational costs, routine cleaning or treatment is therefore a necessity.

Le-Clech et al (2006) did a comprehensive review on sludge characteristics, operational parameters and membrane materials as factors responsible for fine bubble diffuser fouling. The review concluded that no known single parameter can predict or model fouling of fine bubble diffusers, due to changes in biomass characteristics from plant to plant. However an understanding of foulants and their interaction with diffuser polymeric material may provide a new direction for their cleaning and mitigation strategy.

Surfactants and other wastewater characteristic such as COD has also been reported to have adverse effect on OTE and alpha (Rosso et al, 2005; Leu et al, 2009 and Hebrard et al, 2014).

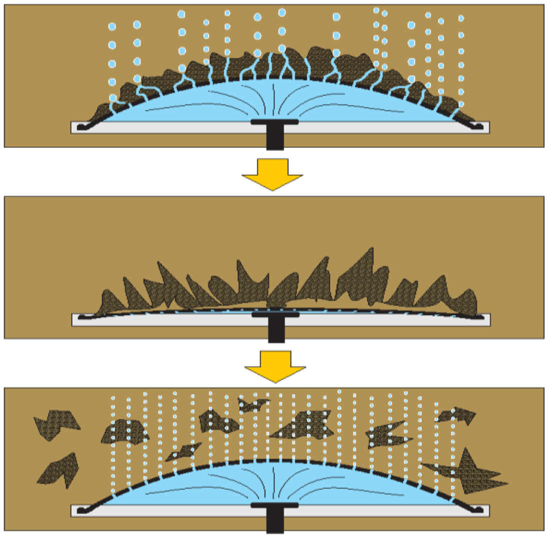
Fine pore diffuser fouling was grouped into three category of bio, organic and inorganic fouling in a previous study (Meng et al, 2009). They defined biofouling as the deposition, growth and metabolism of bacteria's cells or flocs on diffuser membranes. Other studies reported that selected bacteria with higher hydrophobicity than suspended sludge selectively adhere and grow on diffuser surfaces, making their fouling difficult to remove (Jinhua et al, 2006; Miura et al, 2007). Organic fouling was defined as the deposition of biopolymers (i.e., proteins and polysaccharides) on membrane surfaces. The deposited biopolymers were found to be composed of three different layers, a loosely bound cake layer similar to the sludge floc which causes reversible fouling, an intermediate layer with a high polysaccharide content, bacteria aggregates and soluble microbial products (SMP) which causes reversible fouling, and finally a lower layer with high concentration of SMP and proteins which causes irreversible fouling (Metzger et al, 2007). Inorganic fouling, which is the last category type of fouling, was defined as the chemical and biological precipitation of numerous cations and anions. Biopolymers with ionisable groups (i.e. COO^-, CO_3^{2-} SO_4^{2-}, PO_4^{3-}, and OH^-) are easily captured by metal ions (i.e. Ca, Fe, and Mg) present in wastewater. They can form dense cake layers with deposited microbial cells and biopolymers on membrane surface, to cause irreversible fouling (Costa et al, 2006; You HS et al, 2006; Wang et al, 2008). Reversible fouling as described by Meng et al, 2009 is caused by loosely bound foulants on membrane surfaces, and can be remove by physical cleaning like backwashing. Irreversible fouling is caused by fine pore clogging and strongly attached foulant to the membrane surface, due to their affinity for the membrane, and can only be removed through chemical cleaning.

Both physical and chemical cleaning methods have been applied as fouling mitigation technique to fine pore diffusers. Some of the physical cleaning methods applied in previous studies include gas or air sparging, membrane relaxation, pressure and backwashing or flushing (Schiewer and

Psoch, 2005; Schiewer and Psoch, 2006; Le-Clech et al, 2006; Meng et al, 2009; USEPA, 2010). Most manufacturers propose chemical cleaning for fine pore diffuser fouling mitigation, which normally differ in chemical compounds used, concentration and application frequency. In general, the choice of cleaning chemicals depends on wastewater feed characteristics, as acidic cleaning chemicals are most suitable for removing precipitated salt while alkaline cleaning chemicals are suitable for adsorbed organic removal (Van der Bruggen et al, 2003). There is also insufficient information on the effectiveness of the different types of fouling mitigation technique with respect to DWP escalation and OTE decrease respectively.

This study investigates the long-term use of a physical/mechanical cleaning method called reverse flexing (RF), in the mitigation treatment of fine bubble membrane diffusers fouling, its impact on DWP, OTE and alpha factor over a 17 month period. Reverse flexing (RF) is the interruption of air feed and release of pressure from the air feeding line through a constructed vent channel, causing the rapid collapse of the membrane onto the diffuser frame under the action of hydrostatic pressure., Re-inflating the membrane diffuser by opening the air feed aims to scour fouling biofilm, particulates and colloids from the diffuser surface and pores as illustrated in Figure 2.1.

It is therefore hypothesized that the RF treatment method will help to remove both reversible and irreversible fouling, thereby preventing DWP escalation and decrease in OTE and alpha factor. This treatment method is unique and novel because the reactor can be running while the treatment process is going, thereby eliminating the bottle neck of plant shut down and down time as done in other treatment method.



A. Biofilms plugging diffuser pore and surface before reverse flexing

B. Reverse flexing operation

C. Biofilms disengaging from diffuser pore and Surface after reverse flexing

Figure 2.1. Description of the reverse flexing process and the resultant effect

2.5 Materials and Method

2.5.1 *Pilot Reactor Installation*

This study was performed at the Blue Plains advanced wastewater treatment plant, Washington DC, USA. An aeration tank of 60 m^3 volume was constructed in which both clean and process water testing occurred. Four 1 m^2 polyurethane panel diffusers were installed separately in each quadrant of the four quadrants partitioned inside aeration tank with height of 7.62 m and a 3.96 m internal diameter. Clean water test and off gas testing with activated sludge were performed on all four membrane diffuser panels at water depth of 4.88 m inside the aeration tank. Standard oxygen transfer efficiency (SOTE, %) in clean water was measured for all diffusers according to the ASCE standard (2007) protocols. High rate activated sludge (HRAS) pushed through the pilot reactor, was pumped from the first pass of the west secondary reactor in blue plains, one of the largest wastewater treatment facility in the world with a treatment capacity of 1419529.7 m^3/day (375

MGD) . The west secondary reactor treats about 23091 m^3/day (6.1 MGD) of wastewater, and operates at a low DO of 0.1 to 1 mg/ L, sludge retention time (SRT) of 1.5 to 2 days with a targeted average total suspended solid of 2500 mg/L. Low DO in secondary reactors has been proposed by Mueller and Stensel (1990) to favor oxygen transfer. The HRAS from the full scale reactor was recycled to maintain an average HRT and volumetric flow rate of 20 min and 3028 L/min respectively. Air was supplied to the diffusers by a positive displacement blower (Aerzen Blower Model GM 3S). The blower air to the diffusers was regulated at an optimum airflow of 10 m^3/hr per 1m^2 diffuser, by valved acrylic flowmeters (5 – 20 m^3/hr range) supplied by Cole-Parmer. An YSI Pro plus DO meter with probe placed at 50% of the water depth was used to measure DO concentration, while the pilot reactor was operated at an airflow rate of 10 m^3/hr. Since there are no existing guidelines or protocols on the number of times reverse flexing should be done on any diffuser, two of the fine bubble diffusers (RF1 and RF2) were reverse flexed 5 times daily intermittently at 2 – 3 minutes interval, as suggested by the manufacturer and kept constant throughout the 17 months of this study.

2.5.2 *Dynamic Wet Pressure Measurement*

Dynamic wet pressure (DWP) also known as the diffuser headloss or pressure drop, is the pressure differential of a submerged diffusion material normally measured and expressed in inches of water at a specific air flow rate. DWP was measured for the reverse flexed and control diffusers twice a week, using a calibrated Dwyer mercury differential pressure gauge in the range of 0 to 100 inch H_2O (0 to 24 kPa). The differential pressure gauge is design to measure the difference between two pressure points around the diffuser (i.e., the water head (static pressure exerted on the diffuser by the reactor water height) and the blower air inlet pressure to the diffuser), which automatically is the pressure drop across the diffuser. This was to monitor the potential headloss variations caused by fouling over time.

2.5.3 *Oxygen Transfer and Alpha Fouling factor Measurements*

The volumetric mass transfer coefficient (k_La, hr^{-1}) and oxygen transfer efficiency (αSOTE, %) in process water corrected to standard conditions were evaluated for all diffusers according to the ASCE standard 1997 protocols twice a month on all four diffusers, using the off-gas technique (Redmon et al, 1983) and real-time self-calibrating Off-Gas analyzer (Leu et al., 2009). The alpha fouling factor was evaluated as the ratio of process water standard oxygen transfer efficiency over time to clean water oxygen transfer efficiency, to determine the effect of wastewater contaminants and fouling on the fine bubble diffuser transfer efficiency over time (equation 1).

$$\alpha F = \frac{\alpha SOTE}{SOTE} \tag{1}$$

The decrease of aeration performance/efficiency due to fouling of fine pore diffuser was quantified by evaluating the fouling factor (F), which was defined as the ratio of standard oxygen transfer efficiency of the diffuser at any time (t) to standard oxygen transfer efficiency at the initial time (t_0) (equation 2).

$$F = \frac{\alpha FSOTE}{\alpha SOTE} \tag{2}$$

Both DWP and off gas measurements were performed on each diffuser, one at a time, shutting the air supply to the diffusers not being measured in the other quadrants, to prevent any form of air intrusion or interference.

2.5.4 *Sample analysis*

Mixed liquor samples were obtained and analyzed for COD, MLSS and MLVSS after each DWP and off gas measurements using the Hach kits method. Other operating parameters i.e., DO, influent volumetric flow, blower temperature and pressure were also measured. A summary of the pilot reactor operating conditions and average wastewater characteristics, based on samples analysis during each off gas and DWP measurement is reported in (Table 2.1).

Table 2.1. Summary of pilot operational and process condition over the test period

Parameters	Unit	Average Values
Total Influent COD	mg/L	7925±1349
Organic Loading Rate	kgCOD/m3 day	217±40
MLSS	mg/L	6247±1676
MLVSS	mg/L	5236±1453
NH3	mg/L	17±3
NO2	mg/L	0.026±0.003
NO3	mg/L	0.45±0.1
DO	mg/L	0.11±0.08
pH	-	6.78
HRT	min	20±0.4
Standardized Air Flow	m^3/hr	5, 10, 13
Average Volumetric Flow	L/min	3028±110
Temperature	^oC	21±4
Duration	months	17

2.6. Results and Discussion

2.6.1 *Choice of High rate activated sludge system for study*

The choice of a high rate system as feed sludge for this study was to establish the effectiveness of the RF cleaning method in the worst fouling systems. High rate systems are expected to have high mixed liquor suspended solids concentration which inhibits oxygen transfer at the gas to liquid interface. There is also the presence of high concentration of COD and surfactants that also have adverse effect on oxygen transfer efficiency in high rate systems (Hansen et al, 2004; Wagner and Von Hoessle, 2004; Rosso et al, 2007). The pilot feed point in this study was from the first pass in the HRAS reactor where return activated sludge (RAS) from the secondary clarifiers, are fed back into the HRAS reactor to improve treatment and maintain optimum efficiency. The total suspended solid concentration in this section of the HRAS reactor is increased and always higher than the other section of the reactor.

2.6.2 *Quantification of fouling via DWP measurements*

The DWP of both the reverse flexed (RF) and control diffusers did not show any significant change in the first 7 months in operation as shown in Figure 2.2. Only one out of the two control diffusers installed was studied as the one was damaged during installation. An increase in DWP of the control diffuser after 7 months of operation was observed, indicating that fouling of the fine bubble diffuser has started. Previous studies indicated fouling occurrence when no treatment was applied after 12 – 24 months of installation (Rosso and Stenstrom 2006). Given the high risk of fouling in a high rate secondary treatment system, the period indicated to reach fouling in this study was in the expected range (USEPA, 1989; Rosso and Stenstrom, 2006; IWA, 2008; EPRI, 2013). Rosso and Stenstrom (2006) reported that foul diffusers can double their initial DWP over time due to the accumulation of foulants on the surface and pores of the diffusers. The foulant on the diffuser surface thickens over time, reducing its permeability and subsequently increasing its pressure drop. In another investigation of a treated highly alkaline wastewater, the DWP of one of the diffusers increased by 39% of the initial pressure after 3 months and 10 days of operation (Eusebi et al, 2014). In this study about 45 ± 2 % increase in DWP of the control diffuser initial pressure was measured at the end of the 17 months study, which correlates with previous studies (Rosso and Stenstrom 2006a, 2006b). The high increase in DWP of the control diffuser was expected due to high concentration of COD, MLSS and surfactant presence in the HRAS feed used in this study.

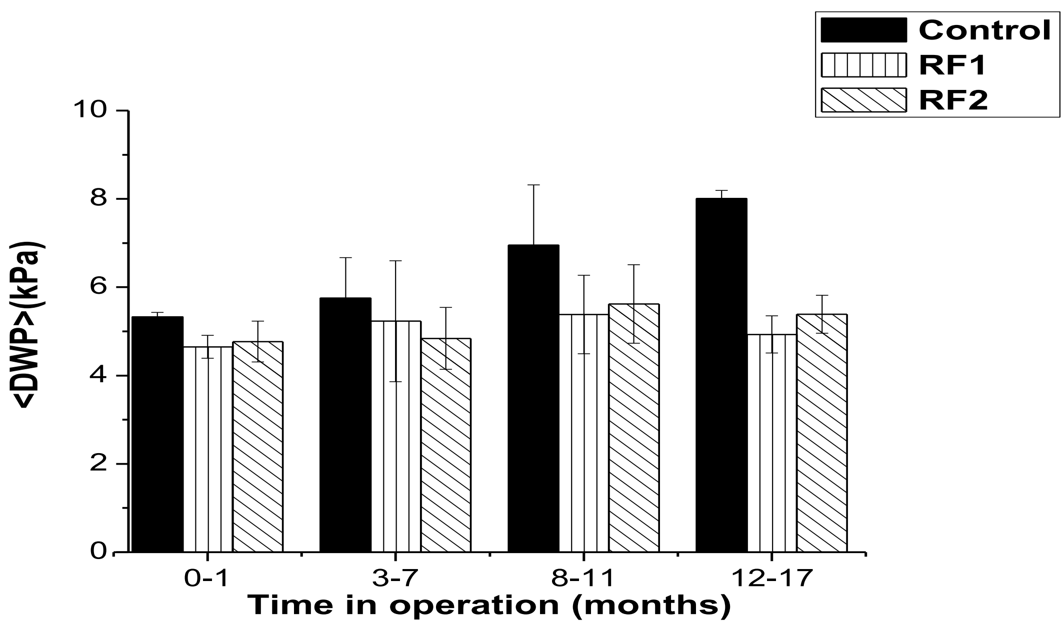


Figure 2.2 Impact of Reverse Flexing (RF) on weighted dynamic wet pressure (DWP) of polyurethane Messner panel diffusers over time in operation

However, no apparent or significant change of DWP was observed between the two RF diffusers during and after the 17 months period in operation as shown in Figure 2.2. A supplementary figure (Figure S1), is provided in the appendix section. This suggested the effectiveness of the RF treatment method in scouring deposited biofilms, colloids and particulates potentially responsible for fouling, from the diffuser surface and blocked pores. The RF treatment method was estimated to have prevented an increase of 35 ± 3 % in DWP for the treated diffusers with respect to the control diffuser. This is a potential pressure burden savings on the blower as described in by Rosso et al, (2008), where a pressure factor P was used to quantify the pressure burden on the blower. When there is excessive pressure burden on the blower as a result of increased DWP caused by fouling, there is a possible air blower compressor surge which inhibits its performance and subsequently damages it, as most air blowers have a designed discharge output pressure which must not be exceeded.

2.6.3 *Impact of Reverse Flexing on αF and F*

A significant decrease in the evaluated α F of both the control and RF treated membrane diffusers was observed after the first 5 months in operation Figure 2.3a. The decline in α F continued over the remaining 12 months in operation but at a slower rate. Rosso and Stenstrom (2006a) reported a high rate of decline in α F (0.5 to 0.33) in the first 24 months of operation, and a slower decline after the 24 months in their study. A result similar to that obtained in this study where α F decrease of 0.55 to 0.35 was observed in the after the first 5 months in operation. Leu et al, 2009 in his study also reported an 18.3 % to 16.3 % decrease in OTE of their installed diffusers after 5 months in operation, which can also be directly correlated to the result obtained in this study. The result in Figure 2.33a did not show any impact of RF in preventing the decrease in α F observed in reverse flexed (RF 1 & RF 2) diffusers, as no apparent difference was observed between the declining α F of the control and RF diffusers over the period of time in operation.

2.6.4 *Potential Reverse Flexing impact limitations on α F and F*

The ineffectiveness of reverse flexing on α F for both RF 1 and RF 2 did not correlate with the DWP measurement observed in Figure 2.2, as one expects a prevention of the continuous decrease in α F for both RF 1 and RF 2, since their DWP increase caused by fouling was shown to be mitigated by the RF process. Diffuser fouling and the consequent increase in DWP as observed in this study impacts most aeration systems with insufficient blower discharge pressure. They bear additional pressure burden (due to DWP increase) caused by fouling, which also produces large bubbles with OTE and α F limitation, because they can only discharge air through diffuser pores with lower head loss (Palm et al, 1980; Rosso et al, 2008). Other studies also reported reduction in diffuser pore permeability caused by blocked pores, leading to an increased pressure drop across the membrane diffuser and large bubble production. This is due to fewer pressurized pore openings which produces large bubbles that decreases oxygen transfer efficiency and α F (Boyle and Redmond, 1983; Kim and Boyle, 1993; Leu et al, 2009). Another important factor, is the fact that over time in operation the diffuser membrane polymeric properties deteriorate by either hardening or softening due to their constant contact with wastewater contaminants and bacteria. This causes an expansion of the diffuser pore size when blocked by biofilms, colloids and particles, which even after treatment, remain unchanged due to their inability to recover from RF stretching. They thereby produce large bubbles with poor oxygen transfer efficiency and lower α F (Hansen et al 2004; Rosso et al, 2007)

Moreover reverse flexing is a physical cleaning method which has also been reported by past studies to remove only reversible fouling caused by loosely bound foulants on fine bubble diffuser surfaces and pores. While irreversible fouling caused by strongly attached inorganic and biofoulant to diffuser surfaces and pores, can only be removed through chemical cleaning (Kim and Boyle, 1993; Le-Clech et al, 2006; Meng et al, 2009).

Therefore, the non-beneficial impact of RF in preventing the decrease in both α F and F as observed in Figure 2.3a and Figure 2.3b, suggests the impact of some level of irreversible fouling of the diffusers. It is also possible that the positive impact of the fouling mitigation in RF1 and RF2 as indicated in Figure 2.2, was not significant enough to offset the wastewater characteristics (MLSS and COD) effect on alpha. As wastewater characteristics and contaminants have been reported by previous studies to have adverse effect on alpha (Rosso et al, 2005; Leu et al, 2009 and Hebrard et al, 2014). The pilot reactor in this study was fed with MLSS from the plants secondary system, which operated with variable MLSS concentration throughout the testing period. There was no evolution trend in MLSS rather a normal variability over time was observed (see Figure S2 in appendix section for additional information). However, measured MLSS and influent variability throughout the study period did not explain evolution of α F with time (see Figure S3 and S4 in appendix section for additional information).

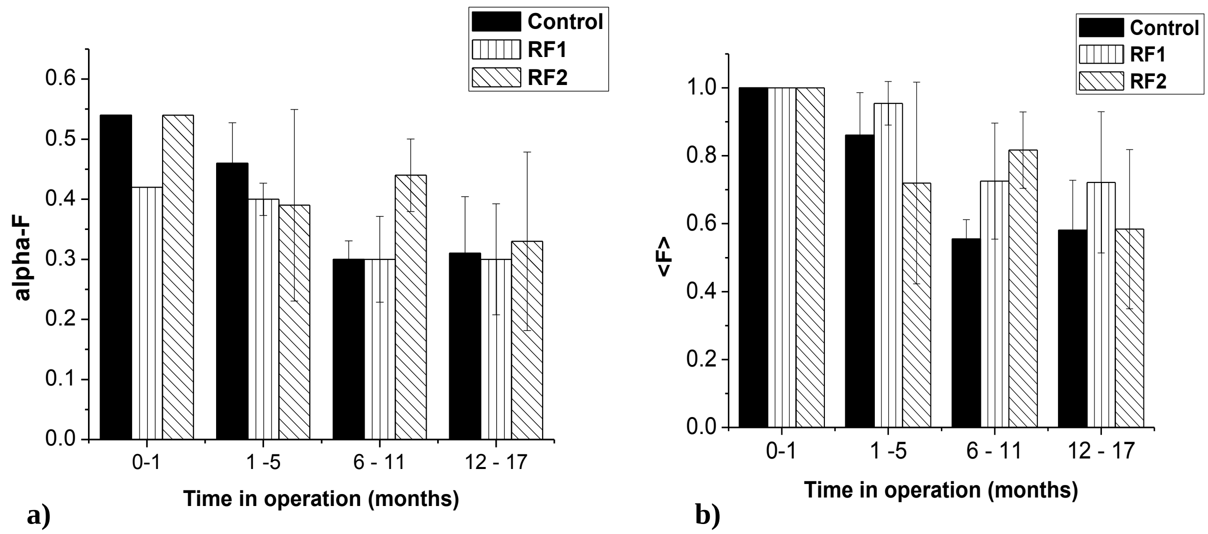


Figure 2.3. Impact of Reverse Flexing (RF) on the evaluated weighted; **a)** alpha fouling (αF) and **b)** fouling factor (F) of the polyurethane messner panel diffusers over time in operation.

2.6.5 *Impact of airflow on standard oxygen transfer efficiency*

The impact of varying the airflow rates on the αSOTE of both the control and RF treated diffusers is shown in Figure 2.4. The result showed no effect of increasing air flow on αSOTE for both the control and RF treated diffusers. The observed result agrees with the Stephenson et al, (2007) report of no change in alpha or αSOTE with increasing air flux. Although other studies have reported contradictory results that show an increase in air flux decreases alpha in fouled diffusers due to fewer pores and slit openings (Stenstrom et al, 2009). However, in this study increasing the air flow did not show any impact on the αSOTE for either the controlled or RF treated diffusers. This is an important information in saving some energy with respect to increasing airflow which increases blower power requirement.

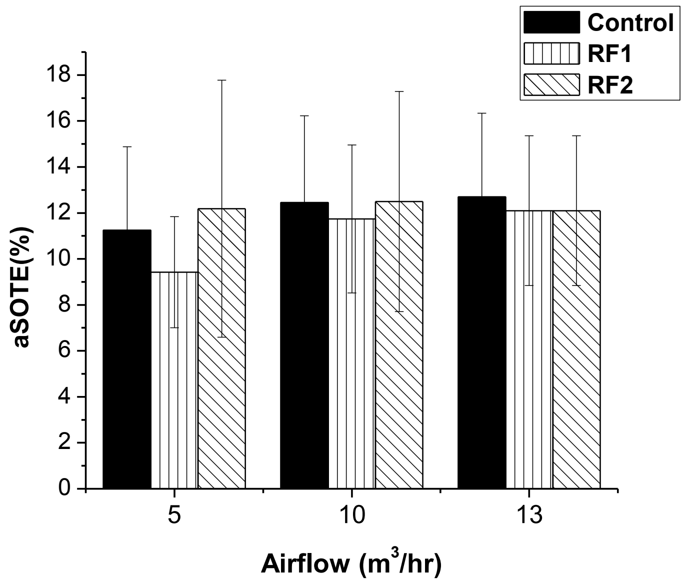


Figure 2.4. Comparing evaluated oxygen transfer efficiency (αSOTE) standardized to process conditions at three different standardized airflow of 5 m3/hr, 6 m3/hr and 8 m3/hr

2.7 Conclusions

Measured DWP indicated the fouling dynamics of fine pore diffusers over time in operation. The applied diffuser fouling physical treatment (RF) method was able to prevent the increase in DWP normally observed in fouling fine pore diffusers over time in operation, saving about 35 % of increase in DWP. The increase in DWP observed in the control diffuser is a potential pressure burden on the blower, which increases the blower energy requirement. However, the RF treatment method did not show any significant impact in preventing the decrease in α F of the fine pore diffusers. Most probably because the RF treatment method could maintained pore openings, remove reversible fouling better but could not fully remove irreversible fouling. There could also have been a dominant impact of other wastewater characteristics on oxygen transfer efficiency and alpha. Further investigation are therefore needed to evaluate the impact of chemical cleaning method that removes reversible and irreversible fouling, this future study will also accommodate oxygen transfer measurements of the foul diffusers in clean water conditions at the end of the study to give an insight on the impact of fouling in alpha fouling decrease. Other future studies include the impact of wastewater characteristics (i.e., surfactants and particulate COD) and operating conditions on OTE and α F, to have a clear understanding on the role of different wastewater characteristics impact on alpha.

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CHAPTER THREE

IMPACT OF WASTEWATER TREATMENT STAGES, WASTEWATER COMPOSITION AND CHARACTERISTICS ON ALPHA AND OXYGEN TRANSFER EFFICIENCY IN THE ACTIVATED SLUDGE PROCESS.

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Prepared for submission to Water Environmental Research.

3.2 ABSTRACT

In this study, the influence of wastewater constituent's removal through different stages of treatment on oxygen transfer efficiency (OTE) and alpha was investigated. The effectiveness of aeration performance with respect to oxygen transfer in the activated sludge treatment process is often defined by the OTE and alpha factor. The alpha Factor (α) is the ratio of standard oxygen transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency of clean water (SOTE). The alpha factor accounts for the wastewater contaminants (i.e. soap and detergent) which have adverse effect on oxygen transfer efficiency. A strong relationship between the wastewater treatment phases and constituents removal in these stages and the alpha factor was established. As expected, increased alphas were observed for the cleanest wastewater matrices (i.e., with increased treated water quality). There was a 46 % increase in alpha as the

total COD and surfactant concentration decreased in the raw wastewater from 300 to 24 mgCOD/L and 12 to 0.3 mgSDS/L in the nitrification/denitrification effluent. The alpha improvement with respect to the different wastewater treatment stages suggested the impact of one or more of the wastewater constituents on OTE and alpha. Batch tests were conducted to characterize the mechanistic impact of the wastewater constituents and found that the major constituents influencing OTE and alpha were surfactants and particulate/colloidal material. The volumetric oxygen mass transfer coefficient (k_La) measurements also showed higher oxygen mass transfer in the nitrification/denitrification effluent compared to raw influent wastewater, complementing the observed alpha results. This study therefore identified the critical wastewater organic constituents impacting OTE and alpha to be surfactants and particulate COD.

3.3 KEYWORDS

Alpha, oxygen transfer efficiency, surfactant, particulate/colloidal COD

3.4 INTRODUCTION

The District of Columbia Water and Sewer Authority which treats about 384 million gallons per day wastewater has a goal to achieve energy-neutral operations in the future. However, operating an energy-neutral municipal advanced wastewater treatment plant presents many challenges. One of the major challenges is to reduce energy consumption through aeration in the carbon and nutrient removal stages because about 45 – 75 % of an average plant total energy cost is incurred during activated sludge treatment (MOP32, 2009; Houck and Boon, 1981; Rosso and Stenstrom, 2005). Aeration has become the most energy and capital intensive aspect of wastewater treatment. In the US as of 1996, the annual electricity use by municipal wastewater treatment plants was 17.4 billion kWh/yr, and in the year 2013 it rose to 30.2 billion kWh/yr, a 74% increase (EPRI and WERF, 2013). The activated sludge treatment process depends on aeration to supply oxygen needed for carbon and nitrogen removal. Activated sludge contains a mixture of microorganisms, bacteria, particles, colloids, natural organic matter, polymers and cations. These activated sludge parameters may impact oxygen transfer efficiency (OTE, %) and alpha, where alpha is the ratio of standard oxygen transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency of clean water (SOTE). Understanding their impact on OTE and alpha can help to improve aeration efficiency and thereby reduce plant operating cost. The volumetric oxygen mass transfer coefficient (K_La) is another parameter which defines aeration performance in the activated sludge treatment process, and it’s a combination of two factors, "K_L" which represents molecular diffusion resistance across the gas and liquid boundary, and "a" which is the surface area. In real wastewater treatment cases, the volumetric mass transfer coefficient K_La is influenced by process parameters like temperature, bubble size, aeration tank depth, air flow rate, liquid depth, tank geometry, and water quality (Hwang and Stenstrom, 1985, Libra, 1993), and is obtained from clean water testing (ASCE, 1993), which can also corrected to standard conditions during wastewater treatment processes.

Several activated sludge characteristics and their impact on OTE and alpha have been investigated in past studies. Duran et al. (2015) in their study, investigated the impact of the mixed liquor suspended solid (MLSS) concentration on K_La in both conventional activated sludge and membrane bioreactor plants. Their study showed a decrease in the K_La with an increase in the MLSS concentration (2.8 to 8.6 g/L). The magnitude of the MLSS impact on K_La was correlated with viscosity which impacts the movement of oxygen molecules and consequently limits their transfer efficiency. Stephenson et al. (2007) investigated different biomass characteristics impact on OTE, where ten biomass samples from municipal, industrial and a submerged MBR pilot plant with a mixed liquor suspended solid (MLSS) concentration in the range of 7.2 to 30.2 TSS g/L at six different air flowrate of 0.7 to 6 m3/m3 /h3 were investigated. Results from that investigation showed an increase in the COD fraction of soluble microbial products (SMP), which decreased alpha and K_La due to surfactants in sludge contributing to the COD. However, Stenstroms (1989) investigation on the impact of biomass MLVSS on OTE in a full scale study from 6 different conventional wastewater treatment plants reported a good correlation between MLVSS and the alpha factor, where the average alpha values at the aeration basin inlet were observed to increase with increasing MLVSS concentration.

In another study of a plug flow aeration system, soluble COD was found to be a principal factor affecting OTE and alpha (Muellar et al., 2000). Leu et al. (2009) in their studies during a 24 hour full scale plant transient test, reported the lowest alpha factor when the COD load was highest, and attributed it to surfactants present in the influent.

Gas transfer processes in liquids are inhibited by surface active contaminants (i.e. surfactants) dissolved in the liquid medium (Rosso et al., 2006). They accumulate at the gas-liquid interface and reduce mass transfer rates. Therefore investigating their effect on OTE and alpha is important in designing and optimizing aeration systems to save energy costs (Rosso, 2005). Wagner et al. (2006) reported in their study that OTE and the alpha factor are a function of sludge type, surfactant concentration and surfactant type. They observed stronger inhibition of OTE and alpha with nonionic surfactants compared to anionic surfactants, although they limited their test to surfactant solutions and a clean water matrix without testing in process water. Two different classes of surfactants (ionics and nonionics) are normally found in wastewater. The ionic surfactants are divided into anionic and cationic. The cationic active portion has a hydrophilic end which is a cation, while the anionic active portion is an anion. Also the nonionic group of surfactants has nonionizable hydrophilic groups which are mostly oxygen, nitrogen or sulfur groups. However, anionic surfactant types make up 73 % of the U .S. consumption of surfactants according to a past study (Rosen, 1978), which is why anionic surfactants were the focus in this study. Hebrard et al. (2014) in their study also reported an enhanced inhibition of mass transfer at the gas-liquid interface by increasing the surfactant concentration. These surfactants congregated and formed a barrier at the gas-liquid interface to prevent mass transfer. Another study (Eckenfelder et al., 1961), also reported that a surfactant (sodium lauryl sulfate) concentration of 15 mg/L in wastewater decreased the oxygen mass transfer coefficient by 50% of the value in clean water tests.

In summary, different sludge characteristics (COD, SCOD, MLSS, MLVSS and surfactants) have been reported to impact OTE and alpha, based on the interfacial mass transfer at the gas to liquid interface and impurities present in the wastewater (Rosso et al., 2005). Soluble COD and its readily biodegradable fraction (rbCOD) has also been observed to reduce alpha under process conditions by OTE suppression (Hwang and Stenstrom, 1983).

The importance of oxygen transfer, microbial activity and metabolism cannot be over emphasized, because the oxygen transfer rate (OTR) in an aerobic process is strongly influenced by the hydrodynamic conditions of oxygen uptake rate (OUR) and K_La in aerobic processes. However, the hydrodynamic conditions depend on operating conditions like sludge retention time, (SRT); air flow; chemically enhanced primary treatment, (CEPT) etc.). Wastewater physicochemical characteristics like readily biodegradable COD (rbCOD), slowly biodegradable

COD (sbCOD) and the presence of oxygen transfer reducing contaminants (i.e., surfactants) are also some of the variables that influence the hydrodynamic conditions of the aerobic process. An indirect effect of OUR on alpha has previously been reported by different studies (Fan et al., 2013; Riber and Stensel, 1985; Bartholomew et al., 1950; Albertson and Digregorio, 1975; Eckenfelder et al., 1979 and Tsao et al., 1968). Reiber and Stensel (1985) reported in their study, that an increasing bulk DO concentration resulted in a decrease of interfacial OTR from the gas to the liquid phase, suggesting a potential effect of low oxygen uptake rate (OUR) on OTR. An increase in OTE with an increasing OUR was also reported by Fan et al. (2013). This can only be possible if there is an alternate direct oxygen transfer pathway by which microbes bypasses the bulk DO and use oxygen directly from the gaseous phase as proposed in some studies (Bartholomew et al., 1950; Albertson and Digregorio, 1975; Eckenfelder et al., 1979).

Adsorption of microbial cells to gas bubbles in the bulk liquid can also increase the interfacial OUR between the microbial cells and gas bubbles, and consequently increase the oxygen transfer rate (Tsao et al., 1968). However, some of these studies have contradicted each other. For example, a decrease in alpha with increasing MLSS and soluble COD was reported by Duran et al. (2015), Muellar et al. (2000), and Leu et al. (2009), while other studies reported either increasing alpha or no change with increasing MLSS or soluble COD (Stenstrom et al., 1989; Stephenson et al., 2007). These conflicting results appear to be due to a lack of knowledge or mechanistic understanding of the impact of wastewater characteristics on alpha and OTE. Aeration design system specification is still marred by both overdesigned and inflated safety factors, since there are no definitive guidelines for alpha prediction with respect to wastewater characteristics and process conditions such as OUR. This study investigated the mechanistic impact of full scale wastewater treatment stages on OTE and alpha. In addition, specific effects of soluble COD, colloidal COD and surfactants on alpha were determined and quantified. The overall goal was to characterize the relationship between wastewater constituents, their concentration, and aeration efficiency. This can be used to prevent the overdesign and consequent energy wastage in activated sludge wastewater treatment.

3.5. MATERIALS AND METHODS

3.5.1 Plant and Pilot Description

This study was carried out in a pilot batch-scale reactor system operated at the Blue Plains Advanced Wastewater Treatment Plant (AWTP) run by the District of Columbia Water and Sewer Authority, Washington DC. The full-scale AWTP treats about 1,458,333 m^3/d (384 million gallons per day) of wastewater. The raw wastewater is first treated by passing it through screens and an aerated grit chamber before it flows into the chemically enhanced primary clarifier. The effluent from the chemically enhanced primary treatment (CEPT) clarifier, is then treated for carbon removal through a biological step-feed high rate activated sludge (HRAS) system. At Blue Plains, the HRAS systems secondary reactors are operated at an SRT of 1.5-2 days, a residual dissolved oxygen (DO) of 0.5-1 mg/L and a targeted average total suspended solid (TSS) of 2500 mg/L. The effluent from the HRAS clarifier flows to the nitrification/denitrification reactor system operating at an SRT of 15 days, for biological nitrogen removal (BNR). Thereafter, an enhanced nitrogen removal (ENR) process is used to remove any residual ammonia still present before it goes to the nitrification/denitrification clarifier. The effluent from the clarifiers is passed through multimedia filtration and disinfection phases before being discharged to the Potomac River.

A 230 L pilot batch-scale cylindrical reactor column with a diameter 0.25 m (10 inches) and a depth of 4.57 m (15 feet) was used in this study. A 9 inch inner diameter ethylene propylene diene monomer (EPDM) fine-pore diffuser disc was installed at the reactor column base (100% coverage). A range of experiments were conducted in the pilot batch-scale reactor system to study a) the effect of the wastewater treatment stages of the AWTP on OTE and alpha, and b) the specific impact of different COD fractions (soluble and colloidal) and surfactants on OTE and alpha. The ultimate goal was to provide a quantitative relationship between the wastewater constituents and the aeration efficiency.

3.5.2 Wastewater treatment stages impact on aeration efficiency experiment

The pilot batch-scale reactor was filled with a mixture of high rate activated sludge (HRAS) and different wastewater processed waters; a) raw influent wastewater (w.wt), (influent w.wt), b) chemically enhanced primary treatment effluent, (CEPT effl), c) carbon removal process effluent and d) nitrification/denitrification effluent, (nit/den effl). The same volume (113.5 L) of each of the treatment stages wastewater was mixed with same HRAS volume (113.5 L) in different sets of experiments, with a final reactor mixture volume and TSS level of 227 L and 100 ~ 110 mg TSS/L, respectively (Table 3.1). The treatment stages wastewater and HRAS mixture characteristics summarized in Table 3.1, are based on completely mixed sample analyses.

The pilot batch-scale reactor mixture was constantly aerated using an air blower compressor which supplied air to the diffuser for 2 hours of aeration in each set of the different treatment stages wastewaters and HRAS mixtures. The airflow was operated at an airflow of 0.28 m3/ hr, by valved acrylic flowmeters (0.05 – 0.5 m3/hr range) supplied by Cole-Parmer. An YSI Pro plus DO meter with a probe placed at 50% of the water depth was used to measure the DO concentrations logged in at 1 min intervals.

3.5.3 OTE and Alpha Estimation

Off-gas testing is a real time process measuring method for estimating air diffuser aeration efficiency defined by OTE and alpha (ASCE, 1997). The off-gas testing method evaluates OTE based on oxygen consumption estimation, by comparing the oxygen content in the supplied air and captured off-gas. During off gas testing, off gas captured by a floating or stationed hood on the surface of an aeration tank or reactor is treated to strip off CO_2 and water vapor, then the oxygen partial pressure is measured by an oxygen analyzer converted to percentage oxygen and analyzed as the mole fraction of O_2.

The percentage of oxygen transferred to the process water is calculated as:

$$\text{OTE (\%)} = \frac{O_{2,in} - O_{2,out}}{O_{2,in}} \qquad \text{(ii)}$$

Where $O_{2,\%}$ = percentage oxygen in the supplied air.

The evaluated OTE is corrected to standard conditions of (20°C, 0 mg $_{DO}$/l, 1 atm, no salinity) to obtain a standardized OTE, or SOTE (%) (ASCE, 1984, 1991). To evaluate the alpha, the SOTE in clean water was measured for the diffuser used in this study according to the ASCE standard

(2006) protocols, and the alpha factor (α) was evaluated by taking the ratio of standardized OTE at process water conditions to clean water condition.

$$\alpha = \frac{\alpha SOTE}{SOTE} \qquad (1)$$

The OTR is also calculated by multiplying OTE by the measured airflow rate passing through the off gas hood.

3.5.4 Sampling and Chemical Analysis

Samples of the batch reactor mixtures were collected at 10 minute intervals during the 2 hours experiment described in sections 3.5.2 and 3.5.3 for each of the different treatment stage wastewaters and the HRAS mixtures. Samples were analyzed for COD fractions and surfactant concentrations (Table 3.1), using a HACH test kit and a HACH DR 2800 spectrophotometer (HACH, Loveland, CO, USA). Total suspended solids (TSS) of the mixture samples (Table 3.1) were also determined using Standard Methods 2540 D and 2540 E, respectively (AWWA et al., 2005). COD fractions were divided into particulate, colloidal, flocculated filtered (soluble) and total COD fractions (pCOD, cCOD, sCOD and tCOD respectively). The sCOD fraction was measured using the flocculation-filtration (ZnSO4 as flocculant, 0.45 μm) method of Mamais et al. (1993) and considered as the true soluble COD fraction. The cCOD was measured as the COD fraction that passed through a 1.5 μm ($COD_{1.5\ \mu m}$) glass-fiber filter (grade 934-AH, Whatman), from which the ffCOD fraction was subtracted (Jimenez et al., 2015). The pCOD was calculated by subtracting COD filtered through 1.5 μm ($COD_{1.5\ \mu m}$) from tCOD (unfiltered sample). Anionic surfactants were measured using the HACH test kit that measured the surfactant as mg/L sodium dodecyl sulphate, using the methylene blue active substances assay (MBAS) method (Chitikela et al., 1995). The MBAS assay, is a colorimetric analysis test method that uses methylene blue to detect the presence of anionic surfactants (i.e., detergent or foaming agent) according to the ASTM International standard technique for detecting anionic surfactants.

Table 3.1 Measured COD fractions, surfactants and TSS of wastewater treatment stages mixed with same high rate activated sludge (HRAS)

Wastewater matrix	Acronym	tCOD mg/L	pCOD mg/L	cCOD mg/L	sCOD mg/L	Surfactants mg/L SDS	TSS mg/L
Raw influent wastwater	**influent w.wt**	288 ± 21	182 ± 11	46 ± 23	62 ± 8	11 ± 1	110 ± 10
Chemically enhanced primary treatment effluent	**CEPT effl**	189 ± 74	95 ± 58	20 ± 2	69 ± 6	8 ± 2	105 ± 8
Carbon removal process effluent	**COD effl**	142 ± 67	121 ± 69	0 ± 0	25 ± 2	1 ± 0.1	100 ± 11
Nitrification/denitrification effluent	**Nit/Den effl**	42 ± 25	21 ± 21	0 ± 0	28 ± 1	0 ± 0.1	100 ± 10

3.5.5 Wastewater characteristics effects on aeration efficiency

To determine the specific effect of COD fractions (soluble and colloidal) and surfactants suspected to have impacts on OTE and alpha, a series of experiments were conducted. To achieve this, different synthetic wastewater mixtures were prepared by adding different concentrations of; i) acetate to achieve CODs of 50, 100 and 200 mgCOD/L as a soluble COD surrogate, ii) 50, 100 and 200 mgCOD/L of powdered potato starch as a particulate/colloidal COD surrogate and iii) 1, 5 and 10 mgSDS/L of sodium dodecyl sulphate as a surfactant surrogate to the same reactor mixture volume of 227 L of HRAS and nitrification/denitrification effluent and TSS (Table 3.2).

Offgas measurements were performed on each set of experiments in i), ii) and iii) to estimate OTE and alpha. Particulate/colloidal COD experiments in ii) were repeated using lower particulate/colloidal COD concentrations (5, 10 and 20 mg/L) using the same powdered potato starch surrogate to show the transition of alpha from lower to higher colloidal COD concentrations. The aim was to determine the minimum particulate/colloidal COD concentration at which a change in alpha is observed.

Table 3.2: Synthetic wastewater mixture (nit/denit effl and HRAS) dosed with different concentrations of various organic contaminants (surrogate) substrate

Substrate Surrogates	Wastewater	Unit	Variables different concetrations used		
			A	B	C
SDS	**nit/denit effl**	**mg/L as SDS**	1	5	10
Potato Starch (PS)	**nit/denit effl**	**mgCOD/L**	50	100	150
Acetate (Ac)	**nit/denit effl**	**mgCOD/L**	50	100	150

*SDS - Sodium Dodecyl Sulfate
All experiments conducted using same synthetic wastewater mixture volume and final TSS concentration of 227 L and 100 mg/L TSS, respectively.

3.6 RESULTS AND DISCUSSION

3.6.1 Impact of Wastewater treatment stages effluent and their characteristics on alpha

In order to determine how each of the different wastewater treatment stages; a) raw influent wastewater, b) chemically enhanced primary treatment effluent, c) carbon removal stage effluent and d) nitrification/denitrification effluent organic constituents impact OTE and alpha, off gas measurements were performed on the HRAS and different wastewater treatment stages mixtures in a series of pilot batch-scale experiments to estimate their OTE and alpha as described in sections 3.5.2 and 3.5.3 The results showed the impact of different wastewater treatment stages on each of their alpha values (Figure 3.1). The raw influent wastewater (influent w.wt) and the chemically enhanced primary treatment effluents matrices had the lowest average alpha values (0.34 and 0.32 respectively) compared to that of carbon removal and nitrification/denitrification stage effluent wastewater alpha values (0.46 and 0.63, respectively). The results showed a strong sensitivity of the average alpha values to the different wastewater treatment stages. In addition, the organic constituents measured in each of the treatment stages also suggested their impact on the measured alpha, as the average alpha summarized in Table 3.3 indicated that raw influent wastewater and chemically enhanced primary treatment effluents had the highest surfactant (11±1 and 8±2 mg/L SDS) and tCOD fraction load (288±21 and 189±74) which could be responsible for their very low average alpha values (0.34 and 0.32).

Table 3.3 Measured COD fractions, surfactants and alpha of wastewater treatment stages mixed with same high rate activated sludge (HRAS)

Wastewater matrix	Acronym	tCOD mg/L	pCOD mg/L	cCOD mg/L	sCOD mg/L	Surfactants mg/L SDS	Alpha α
Raw influent wastwater	**influent w.wt**	288 ± 21	182 ± 11	46 ± 23	62 ± 8	11 ± 1	0.34 ± 0.07
Chemically enhanced primary treatment effluent	**CEPT effl**	189 ± 74	95 ± 58	20 ± 2	69 ± 6	8 ± 2	0.32 ± 0.07
Carbon removal process effluent	**COD effl**	142 ± 67	121 ± 69	0 ± 0	25 ± 2	1 ± 0.1	0.46 ± 0.08
Nitrification/denitrification effluent	**Nit/Den effl**	42 ± 25	21 ± 21	0 ± 0	28 ± 1	0 ± 0.1	0.63 ± 0.06

As there was a 26 % increase in alpha of the carbon removal stage effluent, as the total COD and surfactant concentration decreased from 288±21 to 142±67 mgCOD/L and 11±1 to 1±0.1 mgSDS/L, respectively when compared to the raw influent wastewater. When the nitrification/denitrification stage effluent wastewater was compared to the raw influent wastewater, a 46 % increase in alpha of the former was observed with a decrease in total COD and surfactant concentrations from 288±21 to 42±25 mgCOD/L and 11±1 to 0±0.1 mgSDS/L, respectively. This change in alpha with respect to the different wastewater treatment stages COD concentrations and surfactant concentrations suggest the impact of surfactants or COD fractions on their OTE and alpha values, which is also indicated in the treatment stages strong sensitivity observed. Summarily, Table 3.3 and Figure 3.1 shows that the cleaner the wastewater (low organic constituents) as defined by the treatments stages, the higher the average alpha. This result correlates with past studies (Duran et al., 2015; Muellar et al., 2000 and Leu et al., 2009) which reported low alpha values at high surfactant and COD loadings. Hebrard et al. (2014) also reported inhibition of mass transfer at the gas-liquid interface by an increase in surfactant concentration which congregated and formed a barrier at the gas-liquid interface to prevent mass transfer. Soluble COD and its readily biodegradable fraction (rbCOD) was also reported to have reduced the alpha values under process conditions by suppressing OTE (Hwang and Stenstrom, 1983). Stephenson et al. (2007) in their investigation of the impact of different biomass characteristics on OTE, also observed an increase in the COD fraction of soluble microbial products (SMP), which decreased alpha and k_La due to surfactants in sludge. The volumetric mass transfer coefficient (k_La) was evaluated for all wastewater treatment stages mixtures, which also showed an increasing k_La as the wastewater got cleaner through each treatment stage (Figure 3.5a). Both tests also showed about 45 % decrease in the volumetric mass transfer (k_La) at high surfactant concentration, complementing the decrease in alpha values observed at high surfactant concentrations. Results reported in previous studies were also consistent with the results of this study, showing the impact of COD and surfactant in reducing alpha. However, it is still unclear which of the wastewater characteristics or organic constituents (COD fractions and surfactant) are the primary factor affecting alpha, as changes in each of the organic constituents in the treatment stages shown in Table 3.3 suggest an effect on alpha..

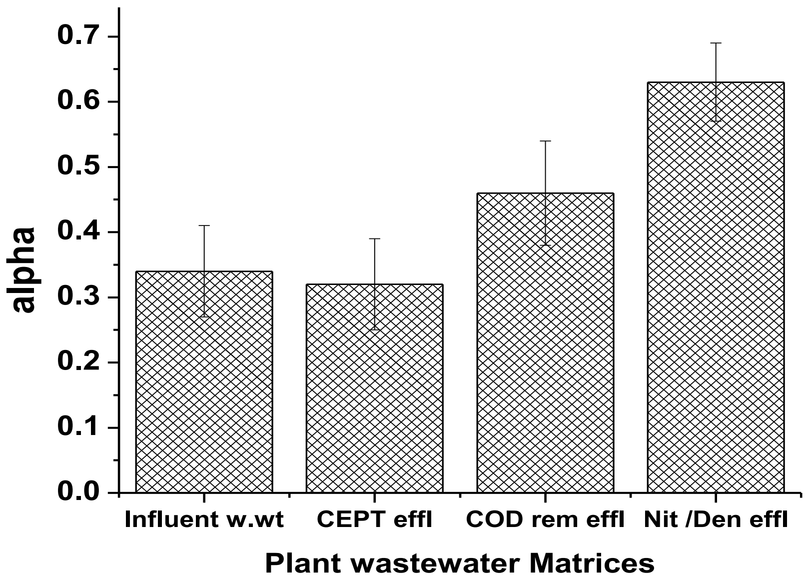

Figure 3.1. Batch test results showing evaluated average alpha factor of Blue Plains different wastewater matrices (Influent waste water (influent w.wt), Chemically enhanced primary treatment effluent (CEPT effl), Carbon removal stage effluent (COD rem effl) and Nitrification/Denitrification stage effluent (Nit/Denit effl)) mixture with High rate secondary activated sludge (HRAS).

3.6.2 Effect of wastewater surfactants, soluble and particulate COD concentrations on alpha

a) Surfactant Impact

In this section the impact of surfactant concentration on OTE and alpha was investigated by performing off gas testing on a set of synthetic wastewater mixtures dosed with different surfactant concentrations in a series of pilot batch-scale experiment described in section 3.5.5. The synthetic wastewater mixture was prepared by dosing nitrification/denitrification effluent to three different concentrations (1, 5 and 10 mg/L) of sodium dodecyl sulphate (SDS) used as the surfactant surrogate, and mixed with HRAS to achieve the same volume of reactor mixture and TSS (HRAS and Nit/Denitrification effluent) described in section 3.5.2. Off gas testing as described in section 3.5.3 was performed on all sets of prepared synthetic wastewater dosed to three different surfactant concentrations to determine their impact on OTE and alpha. All sets of experiments with different surfactant dosages were conducted in triplicate. In addition, off gas testing was also performed on a control (synthetic wastewater without any surfactant). The choice of nitrification/denitrification effluent used to prepare the synthetic wastewater mixture was to minimize any COD interference. Results showed a large decrease in the alpha values as the surfactant concentration dosage was increased (Figure 3.2a). Moreover, the average alpha (0.32) obtained in this section was similar to the alpha values (0.32 / 0.34) observed in the previous experiment for HRAS and the influent wastewater mixture (Figure 3.1), suggesting a direct effect of surfactants on alpha, even at a concentration of 1 mgSDS/L. This result supported the observed alpha increase as the wastewater became cleaner. The anionic surfactants were higher in the raw influent wastewater and CEPT effluent compared to the carbon removal effluent and nit/denit effluent (Table 3.1). The average alpha value of 0.50 obtained in the control reactor mixture (without any surfactant), was also in the range of the alpha values of previously evaluated HRAS and carbon removal effluent / nit/denit effluent wastewaters (0.46 and 0.62, respectively) (Figure 3.1), where COD and surfactant have been removed through treatment.

In another series of experiments to determine the impact of surfactants on OTE and alpha, natural surfactants were harvested from the Blue Plains CEPT effluent channel. The natural surfactants were harvested from the Blue Plains CEPT effluent channel by scooping off foam in the CEPT effluent into a bucket and allowing it to settle into liquid. The harvested natural liquid surfactants were then diluted and used to dose the nitrification/denitrification effluent to the same three different surfactant concentrations of 1, 5 and 10 mg/L measured as sodium dodecyl sulphate, and mixed with HRAS to the same volume of reactor mixture and TSS (HRAS and nit/denitrification effluent) described in section 3.5.2. Off gas testing as described in section 2.3 was also performed on all sets of prepared wastewater mixtures dosed to three different surfactant concentrations to determine their impact on OTE and alpha. All sets of experiments with different surfactant dosages were also carried out in triplicate. Results from this experiment showed similar alpha values (Figure 3.2b) as observed in Figure 3.2a, where synthetic surfactants were added to nitrification/denitrification effluent. The observed result in figure 3.2b also showed a decrease in alpha as the natural surfactant concentration increased. Alpha values in this experiment decreased from 0.50 to 0.34 after 1 mg/L of natural surfactant was added to the reactor mixture. The results in this study also complemented previous studies (Hwang and Stenstrom, 1983; Rosso et al, 2006) that showed that when surfactants accumulate at the gas-liquid interface, they appear to form a layer which causes a reduction in surface tension and gas bubble recirculation, thereby decreasing molecular diffusion into the liquid and reducing the volumetric mass transfer rate. The evaluated k_La result from both tests also showed a 45 % decrease in k_La as the surfactant dosage increased, suggesting an OTR and alpha limitation (Figure 3.5b). Surfactants present as surface active contaminants in the wastewater inhibit gas transfer processes in liquids as they accumulate at the gas-liquid interface where they reduce mass transfer rates.

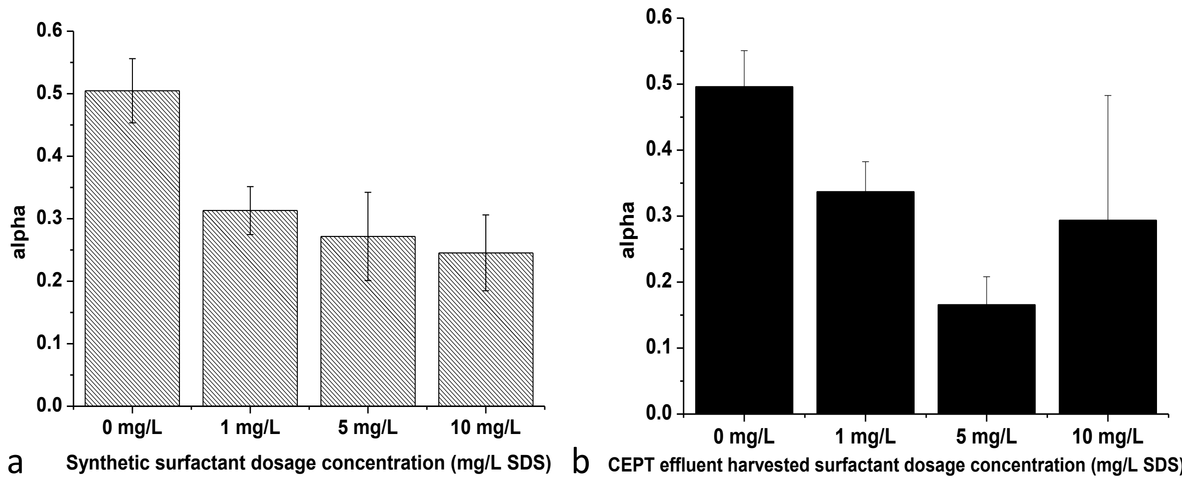

Figure 3.2. Results showing evaluated average alpha factor of spiked a) synthetic and b) harvested CEPT effluent surfactant different concentration to same reactor mixture of HRAS and Nitrification effluent diluted to 100 mg/L TSS

b) Colloidal/Particulate COD impact on alpha

Data in Figure 3.3 shows the impact of particulate/colloidal COD on OTE and alpha. Off gas testing was performed on a set of synthetic wastewater mixtures dosed with different particulate/colloidal COD concentrations in a series of pilot batch-scale experiments described in section 3.5.5. The synthetic wastewater mixture was prepared by dosing nitrification/denitrification effluent to three different concentrations, 50, 100 and 200 mgCOD/L using powdered potato starch and mixed with HRAS to the same volume of the reactor mixture and TSS (HRAS and Nit/Denitrification effluent) described in section 3.5.2. The synthetic wastewater was later dosed with a lower concentrations of powdered potato starch to determine the threshold particulate/colloidal COD concentration above which no change in alpha was observed. Off gas testing as described in section 3.5.3 were performed on all sets of prepared synthetic wastewaters to determine their impact on OTE and alpha. All sets of experiments with different particulate/colloidal COD dosages were conducted in triplicate. In addition, off gas testing was also performed on a control (synthetic wastewater without particulate/colloidal COD). A decrease in the alpha from 0.47 to about 0.27 was observed as the colloidal/particulate COD concentration dosage increased from the initially added 5 mgCOD/L to 20 mgCOD/L (Figure 3). No significant change in alpha was observed above a threshold particulate/colloidal COD concentration of 20 mgCOD/L. Moreover, the average alpha (0.31 and 0.27) obtained at 10 and 20 mgCOD/L in this study was similar to the alphas observed in the previous experiment of surfactant impact on OTE (Figure 3.2a and 3.22b) and the HRAS and CEPT effluent(Figure 3.1). Colloids and particulates can impact OTE and alpha in various ways, from potentially clogging the pores of fine bubble diffusers to increasing the liquid viscosity which impedes the movements of gaseous molecules, thereby reducing the volumetric transfer rate, OTR and alpha (Odize et al., 2017; Rosso et al., 2007 and Leu et al., 2009). Since colloidal and particulate COD biodegrade slowly, there is the possibility of OTR suppression by low OUR which also impacts alpha (Reiber and Stensel, 1985). Results for k_La showed about a 55 % decrease in the volumetric transfer rate with an increase in the colloidal/particulate COD concentration (Figure 3.5c).

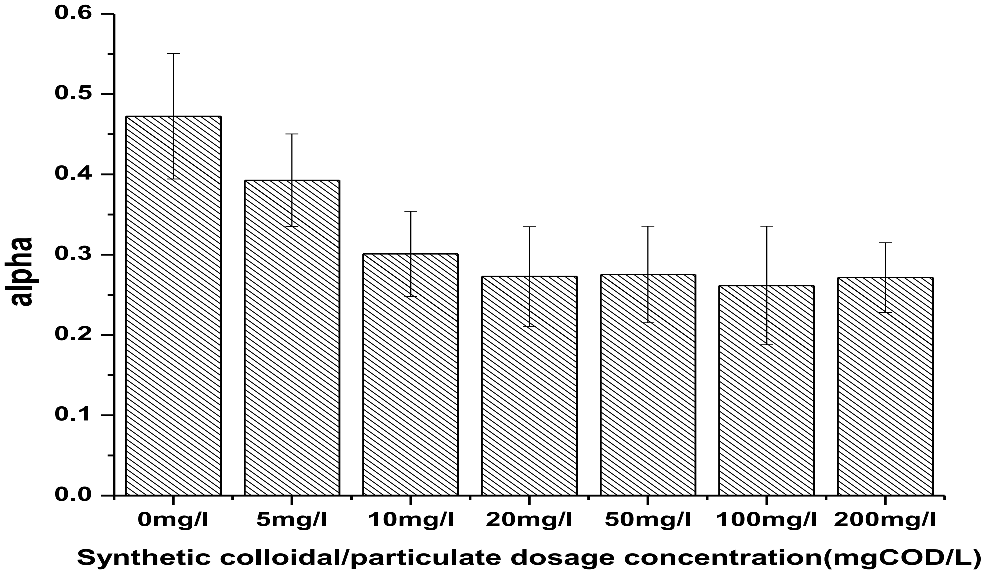

Figure 3.3 Results showing evaluated average alpha factor of a reactor mixture (HRAS + Nitrification effluent diluted to 100 mg/L TSS) spiked with different concentration of synthetic colloidal COD (powdered potato starch)

c) Soluble COD impact on alpha

The results shown in Figure 3.4 summarize the impact of soluble COD on alpha when different concentrations of acetate (soluble COD surrogate) were added to same volume of the synthetic wastewater mixture. The synthetic wastewater mixture was also prepared by dosing nitrification/denitrification effluent with three different concentrations (50, 100 and 200 mgCOD/L) of acetate and mixed with HRAS to the same volume of reactor mixture and TSS described in section 3.5.2. Off gas testing was then performed on all sets of synthetic wastewater mixtures dosed acetate in a series of pilot batch-scale experiments described in section 3.5.5. In addition, off gas testing was also performed on a control. Results showed no apparent change in the alpha, even at increased dosages of soluble COD (Figure 3.4). These results suggest that the soluble fraction of COD has no impact on alpha. However, results in this study contradict other studies that have reported a decrease in alpha with increasing MLSS and soluble COD (Duran et al., 2015; Muellar et al., 2000; Leu et al., 2009). On the other hand, Stenstrom et al., 1989 in their study found a correlation between MLVSS and alpha which showed an increase in alpha with increasing MLVSS concentration. There was also no apparent change in the k_La with increasing soluble COD (Figure 3.5d), which also suggests no impact of soluble COD, in contrast to other studies. The results from this study indicate that the soluble COD fraction in raw influent and CEPT effluent had no impact in OTE and alpha as shown in Figure 3.1.

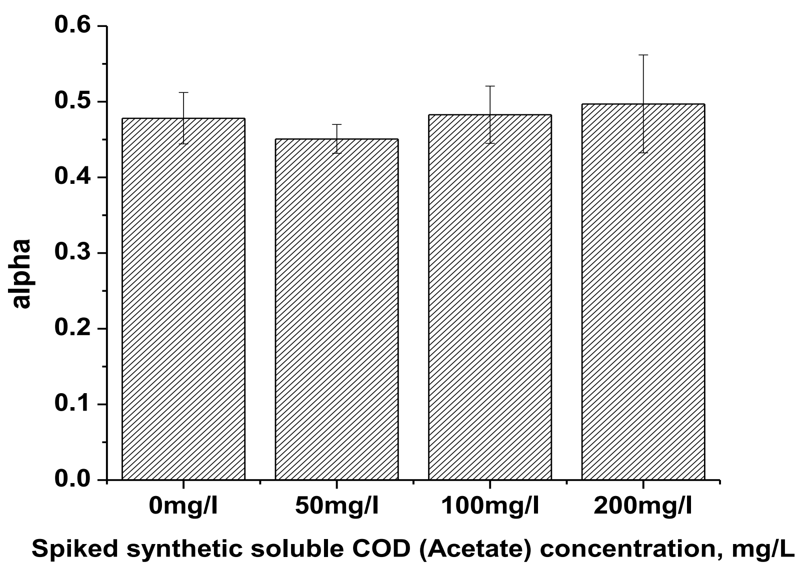

Figure 3.4. Results showing evaluated average alpha factor of a reactor mixture (HRAS + Nitrification effluent diluted to 100 mg/L TSS) spiked with different concentration of synthetic soluble COD (Acetate)

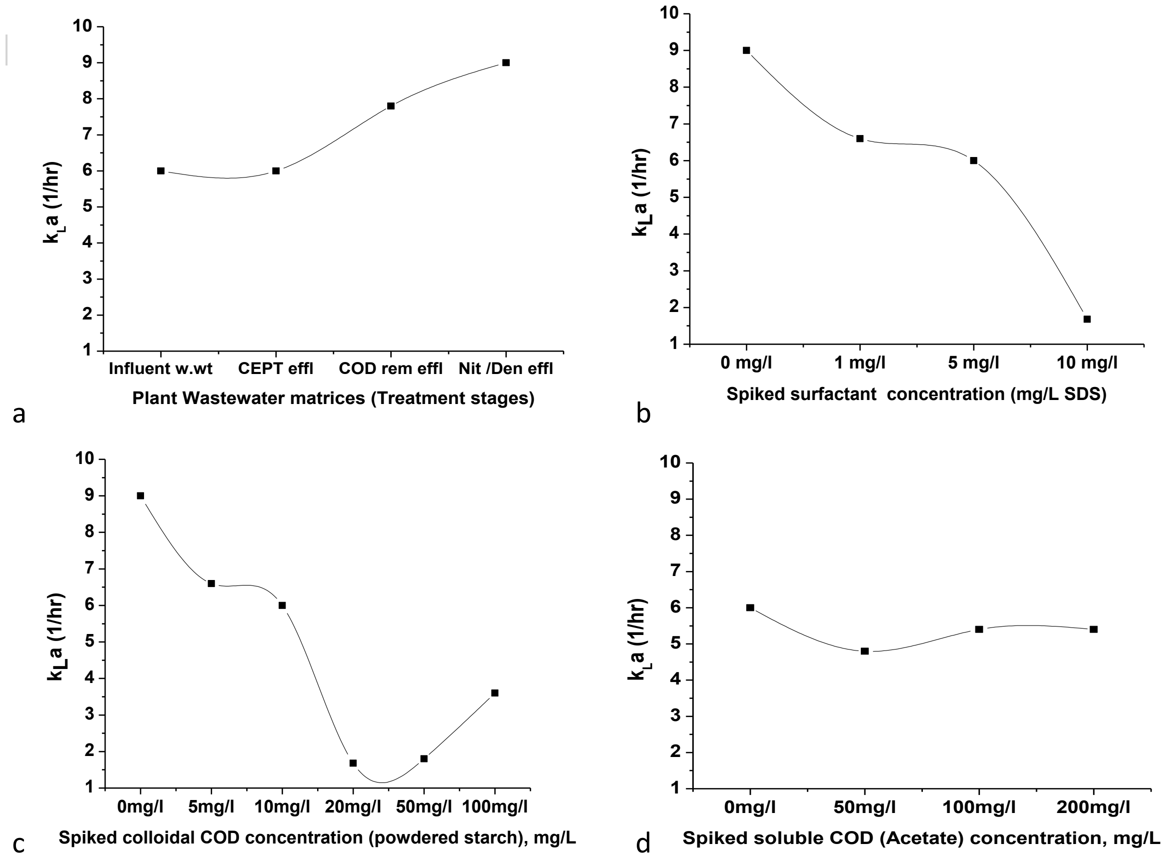

Figure 3.5. Results comparing evaluated k_La of **a)** wastewater matrices added to high rate activated sludge (HRAS), and different concentration of **b)** surfactants, **c)** colloidal COD (powdered starch) and **d)** soluble COD (acetate) added to same reactor mixture of HRAS and Nitrification effluent diluted to 100 mg/L TSS

3.7. Conclusions

A strong relationship between the wastewater treatment stages and alpha was established and as expected increased alphas were observed for the cleanest wastewater defined by its treatments stage and organic constituent (COD and surfactants) measurements. A batch tests conducted to characterize the wastewater organic constituents present in the different wastewater treatment stages impacting OTE and alpha showed that surfactant and particulate/colloidal COD appeared to the critical wastewater organic constituents impacting OTE and alpha. Soluble COD did not have any impact on OTE and alpha. The quantitative mechanistic relationship between the wastewater organic constituent and aeration efficiency in this study, will help to minimize overdesign with respect to aeration system efficiency.

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CHAPTER FOUR

EVALUATION OF STRATEGIES/TECHNOLOGIES ENHANCING SURFACTANT BIOSORPTION AND BIODEGRADATION TO IMPROVE OTE AND ALPHA IN THE ACTIVATED SLUDGE PROCESS

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4.2 KEYWORDS

Alpha, oxygen transfer efficiency, surfactant, contactor stabilization, bioaugmentation, anaerobic selector.

4.3 ABSTRACT

This study was designed to evaluate 3 different activated sludge process treatment strategies/technologies that enhance surfactant removal through sorption and biodegradation and also determine their effect on oxygen transfer and alpha. Pilot and batch scale studies were conducted to compare surfactant removal effectiveness with the alpha of a conventional high-rate activated sludge (HRAS) system, where alpha (α) is defined as the ratio of standard oxygen

transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency of clean water (SOTE). The alpha factor accounts for wastewater contaminants (i.e. surface active substances) which have an adverse effect on oxygen transfer. The 3 enhanced treatment technology were 1) optimized HRAS with contactor-stabilization technology (HRAS-CS), 2) HRAS, bioaugmented (Bioaug) with nitrification sludge (Nit S) and 3) bioaugmented HRAS with an anaerobic selector phase technology (An-S) reactor system configuration. When different stabilization times were evaluated for the optimized HRAS-CS system technology, maximum surfactant removal and alpha were achieved at a 3 hrs stabilization time. For the bioaugmented HRAS technology, the HRAS was bioaugmented with nitrification sludge at different ratios and evaluated. Maximum surfactant removal and alpha were observed at a 50: 50 (HRAS: Nit S) bioaugmentation ratio. Also for the anaerobic selector phase technology, different An-S technology HRTs were evaluated to determine the HRT with the maximum surfactant removal and alpha. A 30 mins An-S HRT was observed to have the highest surfactant removal and alpha. Surfactants in the HRAS-CS system was shown to be removed primarily through enhanced biosorption unto microbial flocs. The treatment technologies evaluated showed maximum surfactant percentage removals of 37, 45, 61 and 87 %, and alphas of 0.37 ±0.01, 0.42 ±0.02, 0.44 ±0.01 and 0.60 ±0.02 for the conventional HRAS, HRAS-CS, Bioaug and An-S reactor system configuration, respectively. The optimized bioaugmented anaerobic selector phase technology showed the highest increased surfactant removal (135 %) through enhanced surfactant biosorption and biodegradation under anaerobic condition, which also resulted in the highest increased alpha (62 %) compared to the conventional HRAS. This study showed that the optimized bioaugmented anaerobic selector phase reactor system configuration is a promising approach to reduce the impact of surfactants on alpha during secondary treatment.

4.4 INTRODUCTION

The biological carbon and nutrient removal processes in an advanced wastewater treatment plants (AWTP) consume about 65 % or more of the plant total energy costs (MOP32, 2009; Houck and Boon, 1981; Rosso and Stenstrom, 2005). To minimize the energy consumption and cost incurred by carbon and nutrient removal stages, new technologies and optimization of existing processes are emerging. To convert these energy intensive processes to less energy intensive ones, high-rate activated sludge systems (HRAS) are getting attention as energy-efficient technology. HRAS systems with different physical and biological optimization processes are now one of the most promising technological approaches to achieve energy neutral wastewater treatment. Oxygen transfer efficiency (OTE) in HRAS systems is the mass of oxygen transferred into the liquid from the mass of air or oxygen supplied, expressed as a percentage (%). While alpha is the ratio of standard oxygen transfer efficiency at process conditions (αSOTE) to standard oxygen transfer efficiency of clean water (SOTE). However, HRAS systems are complex mixtures of microorganisms, bacteria, particles, colloids, natural organic matter, polymers and cations with varying densities, shapes and sizes. These sludge characteristics and components affect OTE and alpha to various degrees due to their impact on mass transfer between the gas and liquid phase (Duran et al., 2015; Rosso et al., 2005; Muellar et al., 2000; Leu et al., 2009). Understanding their impact on OTE and alpha can help to improve aeration efficiency and thereby reduce plant operating cost.

Gas transfer processes in liquids are primarily inhibited by surface active contaminants (surfactants) dissolved in the liquid medium (Odize et al., 2018; Rosso et al., 2006). They accumulate at the gas-liquid interface and reduce mass transfer rates. When surfactants accumulate or are absorbed to the gas bubble surface, there is a reduction in the internal circulation of the gas bubbles. This is because the hydrophobic tail of the surfactant inside the gas bubble creates a stagnant zone which causes a drag effect according to previous literature (Garner and Hammerton, 1954; Clift et al., 1978; Ramirez and Davis, 1999 and Rosso 2005). Both ionic and nonionic classes of surfactants are found in wastewater. The ionic surfactants are divided into anionic and cationic. While the cationic active portion has a hydrophilic end which is a cation, the anionic active portion is an anion. The nonionic class of surfactants have a nonionizable hydrophilic end which is primarily oxygen, nitrogen or sulfur ions.

At the end of the nineteen seventies, more than 80 % of all the surfactants produced were of the anionic type (Wagner et al., 1996). In the United States, about 73 % of surfactants sold are made up of the anionic type (Rosen, 1978). Surfactants are the major ingredients of washing agents, detergents and soap. Due to the vast usage of detergent, soap and washing agents, they are pervasive in wastewater and may be responsible for some of the low oxygen transfer rates in aeration systems (Wagner et al., 1996; Hebrard et al., 2014). The impact of surfactants on OTE in wastewater treatment is normally measured by the alpha factor. The removal of surfactants will thus improve the OTE and alpha which will save energy and cost (Rosso, 2005). Eckenfelder et al. (1961) reported that a surfactant, sodium lauryl sulphate, of 15 mg/L reduced the oxygen mass transfer coefficient to 50 % the value in clean water. Wagner et al. (2006) reported that OTE and alpha were a function of sludge type, surfactant concentration and surfactant type. They found that surfactants are a key factor in the decrease of OTE and alpha. Hebrard et al. (2014) examined the inhibition of mass transfer at the gas-liquid interface by increasing the surfactant concentration, which congregated and formed a barrier at the gas-liquid interface to prevent mass transfer. Rosso et al. (2006) reported the adverse effect of surfactants on the OTE and alpha for both fine and coarse bubble diffusers. Anionic surfactants have been reported to inhibit the removal of slowly biodegradable COD (sbCOD) which is comprised of colloids and particulates. Dereszewska et al. (2015) found that the presence of anionic surfactants above 2.3 mg/L decreased the removal of sbCOD surrogates (starch) over 2 to 3 hrs of experimental time. Dereszewska et al. (2015) also found that under anaerobic condition, a decrease in COD was only observed at the lowest anionic surfactant concentration of less than 2.3 mg/L. When the concentration was increased to 4.2 mg/L they observed that the COD concentration remained unchanged. Othman et al. (2010) showed that anionic surfactants inhibited OUR and nitrification activity. Moreover, it also caused a reduction in sludge flocs size which may lead to poor solids settling in the secondary clarifier.

Surfactant removal to various degrees either through different process technology and operating conditions has been reported by several authors (Mahvi et al, 2004; Rosso et al, 2005; Wagner et al, 2006 and Othman et al, 2010). Surfactant removal through biodegradation in a conventional plug flow activated sludge system was studied by Wagner et al. (2006) and Rosso et al. (2005). Othman et al. (2010), Rosso et al. (2005), and Mahvi et al. (2004) reported surfactant removal under different operating conditions such as a long solids retention time (SRT), bioaugmentation and the use of an anaerobic selector during wastewater treatment. However, there is still limited information about the mechanisms (sorption or biodegradation) of surfactant removal and their impact on OTE and alpha in activated sludge systems.

The goal of this paper was to evaluate 3 different HRAS modification strategies (high-rate contactor-stabilization (C-S), bioaugmentation and anaerobic selector (An-S)) from both process and operating conditions in activated sludge systems to enhance surfactant biodegradation and biosorption during secondary treatment. Several pilot and bench scale batch tests were performed to determine 1) the optimum stabilization time for the CS system, 2) the optimum bioaugmentation ratio for the bioaugmentation strategy and 3) the optimum hydraulic retention time for the anaerobic selector.

4.5 MATERIALS AND METHODS

4.5.1 Plant and Pilot Description

Several pilot and bench scale batch reactor experiments were carried at the Blue Plains Advanced Wastewater Treatment Plant (AWTP) in Washington, DC. The Blue Plains AWTP treats about 1,458,333 m^3/d (384 million gallons per day) wastewater. Plant influent raw wastewater passes through screens and an aerated grit chamber and a chemically enhanced primary treatment (CEPT) stage before flowing into the primary clarifier. A high rate activated sludge reactor system is then used to remove carbon from the CEPT clarifier effluent in the secondary treatment stage. The HRAS systems secondary reactors operate at an SRT of 1.5-2 days, a residual dissolved oxygen (DO) of 0.5-1 mg/L and a targeted average total suspended solid (TSS) of 2500 mg/L. The nitrification and denitrification reactor system which operates at 15 days SRT further treats the HRAS clarifier effluent for biological nitrogen removal (BNR). Residual ammonia and phosphorus still present after the BNR system are removed through an enhanced nutrient removal (ENR) process before flowing into the nitrification/denitrification clarifier. The effluent from the clarifiers is then passed through multimedia filtration and disinfection systems before being discharged to the Potomac River. The bench-scale batch reactor experiments performed in this study were used to determine the optimum stabilization time for maximum extracellular polymeric substances (EPS) production and surfactant removal in the contact stabilization (CS) system under aerobic and anaerobic condition as described in section 4.5.2 and 4.5.3. The pilot scale batch reactor system experiments were used to evaluate 3 different reactor system configurations and strategies that enhance surfactant removal through sorption/biodegradation mechanisms, and their impact on OTE and alpha. The goal was to determine the best reactor system configuration strategy that will improve OTE and alpha through increased or enhanced sorption/biodegradation of surfactants.

4.5.2 Sampling and Chemical Analysis

Samples collected in the series of experiments were also analyzed for COD fractionations and surfactant concentrations using HACH test kits and a HACH DR 2800 spectrophotometer (HACH, Loveland, CO, USA). Total and volatile suspended solids (TSS and VSS) were determined as per the Standard Methods of 2540 D and 2540 E, respectively (AWWA et al., 2005). COD fractionation were divided into particulate, colloidal, flocculated filtered and total COD fractions (pCOD, cCOD, ffCOD and tCOD respectively). The ffCOD fraction was measured using the flocculation-filtration ($ZnSO_4$ as flocculant, 0.45 μm) method of Mamais et al. (1993) and considered as the true soluble COD fraction. The cCOD was measured as the COD fraction that passed through a 1.5 μm ($COD_{1.5\ \mu m}$) glass-fiber filter (grade 934-AH, Whatman), from which the ffCOD fraction was subtracted (Jimenez et al., 2015). The pCOD was calculated by subtracting COD filtered through 1.5 μm ($COD_{1.5\ \mu m}$) from tCOD (unfiltered sample). Anionic surfactants mostly present in the AWTP were also measured using the HACH test kits that measured the surfactant as mg/L sodium dodecyl sulphate (SDS), using the methylene blue active substances assay (MBAS) method. The MBAS assay, is a colorimetric analysis test method that uses methylene blue to detect the presence of anionic surfactants (i.e., detergent or foaming agent) according to the ASTM International standard technique for detecting anionic surfactants.

4.5.3 EPS measurement

Extracellular polymeric substances were measured in the contactor and stabilizer reactor from samples collected from the batch and pilot-scale experiments. The heat extraction method of Li and Yang was modified to extract the EPS from activated sludge. A 30 mL volume of fresh sludge sample was centrifuged at 4000 g for 5 minutes. After discarding the supernatant, the sludge pellets were then resuspended in 10 mL of heated (at 60 ^{o}C) phosphate buffer solution (PBS). The PBS solution consisted of 2 mM (0.328 g/L) Na3PO4, 4 mM KH2PO4 (0.544 g/L), 9 mM NaCl (0.525 g/L) and 1 mM KCl (0.074 g/L). The PBS solution's pH was adjusted to 7.2 and it was kept at 60 ^{o}C before EPS extraction for resuspension. After resuspension of the sludge pellet in 10 mL PBS, the sample was immediately vortexed for 1 minute and then centrifuged at 4000 g for 10 minutes. The organics contained in the supernatant after centrifugation measured as COD, were considered as the loosely-bound (LB) EPS. The remaining pellet was again resuspended in 10 mL of PBS solution and digested for 30 minutes at 60 ^{o}C. After digestion, the sample was centrifuged at 4000 g for 15 minutes. The supernatant of the sample after centrifugation considered as tightly-bound (TB) EPS. The COD of the supernatant after LB-EPS and TB-EPS extraction were measured as mgCOD/L and then corrected by dividing the ratio of the volume of sludge sample over the volume of PBS solution (30 mL/10 mL = 3.0). The summation of LB-EPS and TB-EPS provides the total EPS and calculated as mgCOD/gVSS unit using the sludge sample VSS concentration.

4.5.4 OTE and Alpha Estimation

Off-gas testing is a real time measuring method for estimating air diffuser aeration efficiency defined by OTE and alpha (ASCE, 1997). The off-gas testing method evaluates OTE based on oxygen consumption estimation, by comparing the oxygen content in the supplied air and captured off-gas. During off gas testing, off gas captured by a floating or stationed hood on the surface of an aeration tank or reactor column is treated to strip off CO_2 and water vapor, then the oxygen partial pressure is measured by an oxygen analyzer converted to percentage oxygen and analyzed as the mole fractions of O_2.

The percentage of oxygen transferred to the process water is calculated as:

$$\text{OTE (\%)} = \frac{O_{2,in} - O_{2,out}}{O_{2,in}} \qquad \text{(ii)}$$

Where $O_{2,\%}$ = percentage oxygen in the supplied air.

The evaluated OTE is corrected to standard conditions of (20°C, 0 mg $_{DO}$/l, 1 atm, no salinity) to obtain a standardized OTE, or SOTE (%) (ASCE, 1984, 1991). To evaluate the alpha, the standard oxygen transfer efficiency (SOTE, %) in clean water was measured for the diffuser used this study according to the ASCE standard (2006) protocols, and the alpha factor (α) was evaluated by taking the ratio of standardized OTE at process water conditions to clean water condition.

$$\alpha = \frac{\alpha SOTE}{SOTE} \qquad (1)$$

The oxygen transfer rate (OTR) is also calculated by multiplying OTE by the measured airflow rate passing through the off gas hood.

4.5.5 Effect of stabilization time on extracellular polymeric substance (EPS) production and surfactant removal in the contact stabilization (CS) system

In an HRAS-CS system, the stabilization time is a vital parameter as it creates the needed starvation (famine) condition for microorganisms to maximize EPS production and biosorption of surfactants in the contactor (Rahman et al, 2017). Therefore a determination of the optimum stabilization time needed to enhance the performance of the HRAS-CS configuration will help to reduce excessive energy consumption and save cost. To determine the optimum stabilization time for maximum EPS production and surfactant removal, several bench-scale batch experiments at different stabilization times (0, 3, 6 and 15 hrs) were performed under aerobic conditions in the HRAS-CS system (Table 4.5a).

Table 4.5a: Contactor-Stabilization batch experiment operational parameters and variables

Variables	Units	Average values
Stabilization time	Hrs	0, 3, 6 and 15
Reactor Volume	L	2
Reactor (Contactor) mixture volumetric ratio	HRAS : CEPT effluent	50 : 50
Contactor time	Hrs	1
Contactor Total COD (tCOD)	mgCOD/L	234±11
Contactor Soluble COD (sCOD)	mgCOD/L	110±4
Contactor Surfactant	mgSDS/L	7.2±1
Contactor TSS	mg/L	110±10

The HRAS in the Blue Plains Advanced Wastewater Treatment Plant, Washington, DC, where the sludge sample was collected is presently bioaugmented with nitrification/denitrification sludge in a ratio of 50:50. About 50 % of the waste activated sludge (WAS) from the biological nutrient removal (BNR) or nitrification/denitrification stage is recycled to the HRAS. Two liters of the bioaugmented HRAS sludge was collected from the full-scale plant HRAS reactor and aerated for 3 hrs in a beaker at a DO ≥ 6 mgO_2/L condition without the addition of any substrate. This is to mimic starvation conditions in the Full-scale system. At the end of the 3 hr stabilization period, the air was turned off and 1 L of the stabilized sludge was removed and replaced with 1 L of CEPT effluent to simulate the contactor reactor phase. The reactor sample mixture was reaerated to keep the DO in the contactor phase above 2 mgO_2/L for carbon oxidation and to maintain aerobic conditions for 1 hr. Mixing of the reactor sample mixture with a magnetic stir bar and stirrer plate was also continuous throughout the experiment. Samples were collected at end of the stabilization period, and at 5, 30 and 60 min in the contactor phase, respectively. In each sample, COD fractions (tCOD and SCOD), TSS, and EPS fractions (Loosely Bounded (LB-EPS) and TB-EPS) were measured.

The experiment was also repeated for stabilization times of 360 mins (6 hrs) and 900 mins (15 hrs) as shown in Table 4.5a .A controlled experiment without HRAS pre-aeration (0 hr stabilization time) was also carried out by adding 1 L of the HRAS into the contactor phase with 1 L CEPT effluent. This was to compare the conventional HRAS configuration to the CS system configuration at different stabilization times. The contactor reactor phase mixture average initial organic constituents composition measured are shown in Table 4.5a.

4.5.6 Determination of surfactant removal mechanism in HRAS-CS

The experiment in section 4.5.5 was repeated under anaerobic conditions with the goal of comparing surfactant removal mechanisms (sorption or biodegradation) under both aerobic and anaerobic conditions, as the surfactant would not be expected to be biodegraded in the absence of oxygen (Rahman et al, 2016 and Kinyua et al 2017). To create anaerobic conditions, following aeration, the air was turned off at the end of the different stabilization times and nitrogen gas was use to purge the reactor of DO to create anaerobic conditions in the reactor. Thereafter, 1 L of the stabilized sludge was removed and replaced with same volume (1 L) CEPT wastewater in the contactor reactor. The contactor phase reactor sample mixture was then continuously mixed using a magnetic stir bar and stirrer plate, while constantly supplying nitrogen to maintain anaerobic conditions in the contactor phase for the 1 hour of the experiment. A controlled experiment without HRAS pre-aeration (0 hr stabilization time) was also carried out by adding 1 L of the HRAS into the contactor phase with 1 L CEPT effluent, and also maintaining anaerobic conditions by supplying nitrogen for the 1 hr contact phase. This was also to compare the CS system configuration to the conventional system at different stabilization times.

4.5.7 Evaluating the impact of the CS system configuration on OTE and alpha

A pilot-scale batch reactor was used to evaluate the impact of the CS system reactor configuration on surfactant removal and OTE/alpha. The pilot-scale batch reactor consisted of a 230 L cylindrical reactor column with diameter of 0.25 m and depth of 4.57 m. A 0.23 m inner diameter EPDM fine-pore diffuser disc is also installed at the reactor column base (100% coverage). 113 L of the bioaugmented HRAS sludge was collected from the full-scale plant HRAS reactor and aerated for 3 hrs in a 230 L pilot reactor column at a DO ≥ 6 mgO_2/L condition without the addition of any substrate. This is to mimic starvation. At the end of the 3 hr stabilization period, the air was turned off and 113 L of CEPT wastewater was added to the pilot reactor column to mimic the contactor reactor phase. The contactor reactor mixture was reaerated to keep the DO above 2 mgO_2/L for carbon oxidation and to maintain aerobic conditions for the 1 hr of the contact phase. Off gas measurements as described in section 4.5.4 were performed on the contact phase to evaluate the OTE and alpha of the CS system reactor configuration. An air blower compressor was used to supply air to the diffuser for the 3 hour stabilization phase and the 1 hour contactor phase of aeration in the experiments. The contactor reactor column was covered with a lid to create the headspace for offgas capture needed to evaluate the OTE and alpha. The airflow was regulated at an optimum airflow rate of 0.28 m^3/ hr, by valved acrylic flowmeters (0.05 – 0.5 m3/hr range) supplied by Cole-Parmer. An YSI Pro plus DO meter with the probe placed at 50% of the water depth was used to measure DO concentrations logged in at 1 min intervals. Offgas measurements were also performed on a controlled experiment using the conventional HRAS configuration with only CEPT wastewater and HRAS (without pre-aeration) contactor phase. This was to compare the conventional plug flow HRAS configuration to the CS system. Both experiments were repeated under anaerobic conditions described in section 4.5.6 in a 230 L reactor column without offgas testing, to serve as the sorption control system. Samples were collected at end of the stabilization period, and at 5, 30 and 60 min in the contactor phase, respectively. In each sample, COD fractions, TSS, and EPS fractions were measured. The difference between EPS (standardized by VSS concentration) of the contactor (EPS_C) and EPS of stabilizer (EPS_S) was divided by the EPS of stabilizer and expressed as % of relative EPS increase or decrease in the contactor.

4.5.8 Evaluating the impact of the bioaugmentation system configuration on OTE and alpha

At Blue Plains, each of the HRAS secondary reactors treats about 6.1 MGD of wastewater, and operates at a SRT of 1.5 to 2 days with a targeted average total suspended solid of 2500 mg/L. The secondary reactors are separated into west (bioaugmented) and east (non bioaugmented) secondary reactors. Both reactors are installed with coarse bubble diffusers which operate in parallel with a residual DO of 0.5 to 1 mg/L. About 50 % of the waste activated sludge (WAS) from the biological nutrient removal (BNR) or nitrification stage is recycled to the west reactor, while the non bioaugmented reactor operates normally without receiving any nitrification/denitrification sludge. Several pilot-scale batch experiments were conducted to determine the optimum bioaugmentation ratio system configuration with the highest improved OTE and alpha with respect to surfactant removal. The 230 L pilot-scale batch reactor was filled with a mixture of chemically enhanced primary treatment effluent and bioaugmented high rate activated sludge (HRAS) in a volumetric ratio of 50:50 and final reactor mixture TSS level of 100 ~ 110 mg TSS/L (Table 4.5b). The HRAS sludge was bioaugmented at 4 different ratios of nitrification/denitrification sludge in 4 different set of experiments as shown in Table 4.5b.

Table 4.5b: Bioaugmentation reactor system configuration experiment operational parameters and variables

Variables	Units	Average values
Bioaugmented sludge ratio	HRAS : Nit/Denit Sludge	100 : 0, 0 : 100, 50 : 50 and 25 : 75
Reactor Volume	L	230
Reactor mixture volumetric ratio	HRAS : CEPT effluent	50 : 50
Experiment duration	Hrs	1
Total COD (tCOD)	mgCOD/L	234±11
Soluble COD (sCOD)	mgCOD/L	110±4
Surfactant	mgSDS/L	5.2±1
TSS	mg/L	110±10

Off gas measurements as described in section 4.5.7 were performed on all 4 bioaugmented configurations to evaluate their OTE and alpha with respect to surfactant removal.

4.5.9 Evaluating the impact of adding an anaerobic selector to the bioaugmented system reactor configuration with respect to OTE and alpha.

The first zone in each HRAS reactor at Blue Plains serves as an anoxic or anaerobic selector zone with a 45 min hydraulic retention time (HRT) when operating in the bioaugmentation mode. The anaerobic selector prevents filament proliferation and promotes development of biomass with increased sorption capacity and good settling characteristics (Albertson, 2005). The HRT is thought to be a critical operational parameter. The anaerobic selector is expected to reduce soluble COD to the range of the final effluent in 12 to 30 minutes on raw or settled effluent according to a previous reports (Albertson, 2005; Azimi and Zamanzadeh, 2006). In this study, experiments were performed to determine a) the impact of adding an anaerobic selector to the bioaugmentation system reactor configuration and b) the effect of varying anaerobic selector HRT with respect to OTE, alpha and surfactant removal. The 230 L pilot-scale batch reactor was filled with a mixture of chemically enhanced primary treatment effluent and bioaugmented high rate activated sludge (HRAS) in a ratio of 50 : 50 with final reactor mixture TSS level of 100 ~ 110 mg TSS/L (Table 4.5c).

Table 4.5c: Anaerobic selector reactor system configuration experiment operational parameters and variables

Variables	**Units**	**Average values**
Anaerobic Selector HRT	Min	15, 30 and 45
Reactor Volume	L	230
Reactor mixture volumetric ratio	Bioaugmented Sludge : CEPT effluent	50 : 50
Experiment duration	Hrs	1
Total COD (tCOD)	mgCOD/L	234±11
Soluble COD (sCOD)	mgCOD/L	110±4
Surfactant	mgSDS/L	5.0±1
TSS	mg/L	110±10

A 15 min HRT anaerobic selector phase was created by purging the reactor sample mixture with nitrogen gas to maintain DO at 0 mg/L for 15 min. Thereafter air was added for another 1 hr of the experiment to create aerobic conditions while also performing off gas measurements as described in section 4.5.7. The experiments were repeated for an anaerobic selector times of 30 and 45 min (Table 4.5c).

4.6. RESULTS AND DISCUSSION

4.6.1 Determination of the optimum stabilization time for maximum EPS production and surfactant removal for an aerobic HRAS-CS system.

To determine the optimum stabilization time to maximize EPS production in the contactor, the HRAS-CS technology was simulated in a batch reactor using stabilization times of 0, 3, 6 and 15 hrs using the HRAS with 1.5 to 2 d SRT from plant full-scale systems as described in section 4.5.5.

Results in Figure 4.1a showed that the EPS concentration remained unchanged during the 180 min (3 hr) stabilization period and increased during the first 5 min of contactor time following addition of CEPT effluent. The COD increased from an initial 244 mgCOD/gVSS to 298 mgCOD/gVSS. A similar result was reported by Rahman et al, 2017, who suggested that microbes transition very quickly from a low carbon oxidation rate in the stabilizer to a high carbon oxidation rate in the contactor, which potentially enhances EPS production in the aerobic contactor as was also observed in this study. The EPS concentration was observed to decrease after 5 mins of contactor time, suggesting a possible decrease in EPS production combined with EPS consumption due to a carbon limitation. Because the aerobic microorganisms consume EPS as a substrate and they cannot replenish EPS due to a carbon limit, a decrease in EPS will occur. This result also complements a study by Rahman et al, 2017 who reported EPS degradation beyond 5 min of contact time in a CS system. Results shown in Figure 4.1b compared the EPS in the 180 min stabilization time HRAS-CS experiment to that of the conventional HRAS system without stabilization, where a 67 % increase in EPS production in the contactor was observed in the latter system.

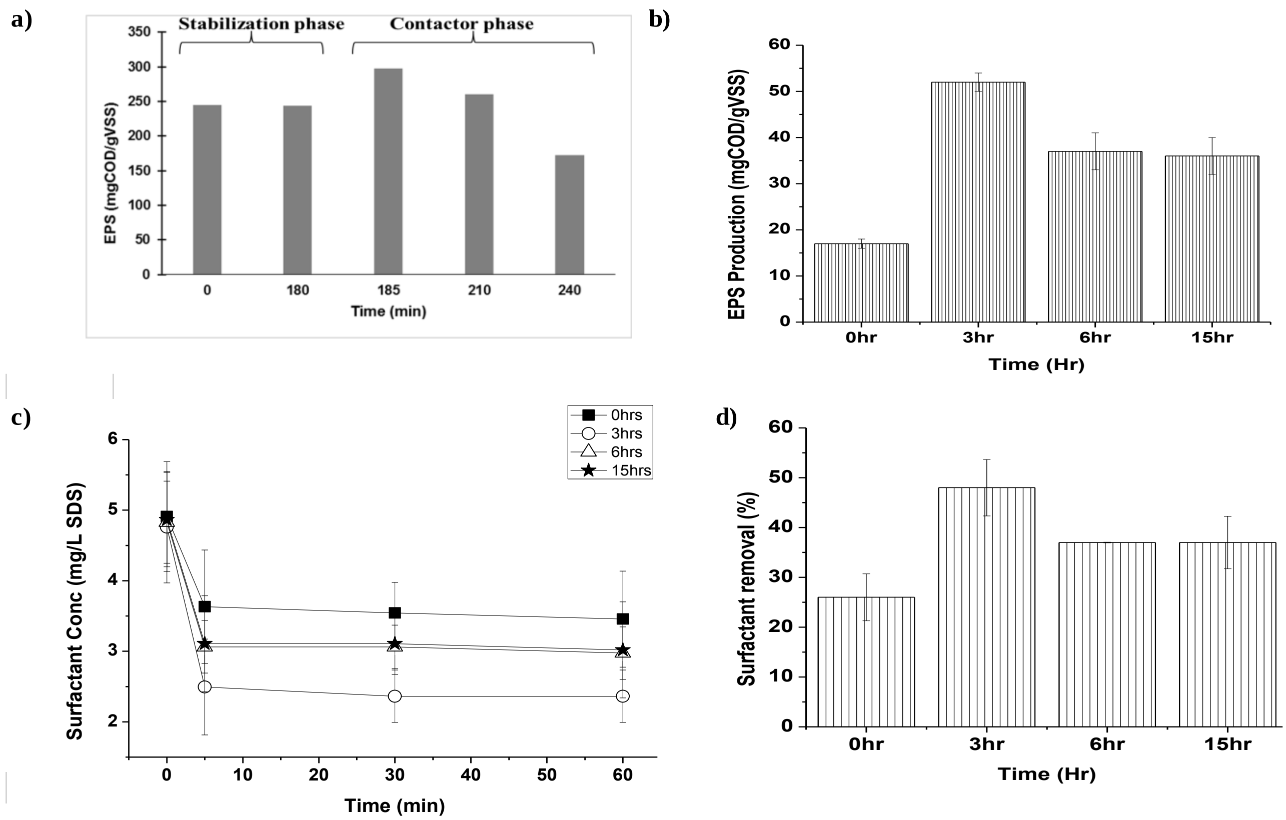

Fig 4.1. (a) Specific EPS concentration over 3 hrs stabilization and 1 hr contactor time, **b)** Specific EPS concentration at 5 mins of aerobic contactor time after each of the stabilization time, **(c)** Surfactant concentration profile over 1 hr aerobic contactor time after each of the stabilization time **(d)** Percentage surfactant removal over 1 hr aerobic contactor time after each of the stabilization time

These results suggests the impact of stabilization time on EPS increased production in the contactor, which possibly enhanced biosorption of surfactants (Figure 4.1d). Past studies (Rahman et al., 2016; Jenkins and Orhon, 1973 and Siddiqi et al., 1965) have reported that during the stabilization phase, microorganisms release some exo-enzymes that help to hydrolyze and solubilize material sorbed on the microbial flocs. The enzymes are returned back into the contactor which helps to accelerate substrate utilization which increases EPS production, hence the increase in EPS production observed in the 180 mins HRAS-CS test compared to the conventional HRAS test in this study.

Results shown in Figure 4.1b also compared the produced EPS in the 180 min stabilization time HRAS-CS experiment to that of the 360 mins (6 hrs) and 900 mins (15 hrs) stabilization time

HRAS-CS experiment, and a decrease in EPS production in the latter was observed. The goal was to determine the optimum stabilization time for maximum EPS production, since stabilization time was observed to be vital in EPS production. The decrease in EPS for the higher stabilization times of 360 and 900 mins compared to 180 min using same 1.5 – 2 day HRAS sludge suggests a decline in the fraction of active biomass still present in the HRAS. This is because starvation of microbes at longer stabilization times can lead to cell lysis, causing a decrease in the active microbial fractions. Rahman et al., 2017 in his study reported an increased EPS production in the contactor for an 86 mins stabilization time experiment compared to a 104 mins stabilization time experiment, using sludge in an SRT range of 0.52-0.75 d. A higher specific OUR obtained for the 86 mins stabilization time experiment compared to 104 mins in the study, when acetate was spiked in both stabilizer sludge, further confirmed the possible decrease in active microbial biomass in the 104 mins stabilization time experiment .

In Figure 4.1c, the surfactant concentration profile over the 1 hour contact time showed that most of the surfactants were removed in the first 5 mins in the contactor, and no change in surfactant concentration was observed beyond the 5 mins of contact time. Results in Figure 4.1d also showed that a 46 % increase in surfactants removal was observed for the 180 mins stabilization time HRAS-CS test when compared to the conventional HRAS (0 stabilization time) test.A 30 % increase in surfactants removal was observed when the 180 mins stabilization time test was compared to the 360 and 900 min stabilization times test in the HRAS-CS. The highest percentage surfactant removal observed for the 180 mins stabilization time HRAS-CS test complements its highest EPS production observed in Figure 4.1b compared to the conventional HRAS, 300 and 900 mins stabilization time HRAS-CS system. The results in Figure 4.1b and 4.1d also suggests that an increase in EPS production can potentially enhance surfactant removal, and a 180 mins (3 hr) stabilization time is the near optimum stabilization time for maximum EPS production and surfactant removal for the optimized HRAS-CS system. The results in this study agree with a past study which also reported an optimum 180 min stabilization time for maximum EPS production in a 2.5 d SRT HRAS (Rahman et al, 2017).

4.6.2 Determination of surfactant removal mechanisms (biosorption/biodegradation) under aerobic and anaerobic conditions.

A series of batch-scale experiment were conducted to determine the surfactant removal mechanism in the different HRAS systems as described in section 4.5.5 and 4.5.6. The results represented in Figure 4.2a showed no significant difference in surfactant removal for both aerobic and anaerobic reactor conditions in the HRAS system. This result suggests that surfactant removal under the aerobic conditions is primarily through sorption, since no significant improvement in surfactant removal was observed under aerobic conditions when compared to anaerobic condition where surfactants are not thought to biodegrade, but only adsorb. An increased surfactant removal should have been observed under aerobic conditions if the surfactant removal through biosorption and biodegradation were combined.

When same experiment was repeated under an optimized HRAS-CS system configuration with a 3 hrs stabilization time, there was an increase in surfactant removal under both aerobic and anaerobic conditions compared to the conventional HRAS test. However, there was still no difference in surfactant removal between the aerobic HRAS-CS test and the anaerobic HRAS-CS test. This result is similar to results for the conventional HRAS results, suggesting that surfactant removal in the optimized aerobic HRAS-CS system was primarily through sorption, and that the increase in surfactant removal observed when compared to the conventional HRAS system was the result of a stabilization effect which enhanced surfactant sorption capacity, as discussed in the previous section.

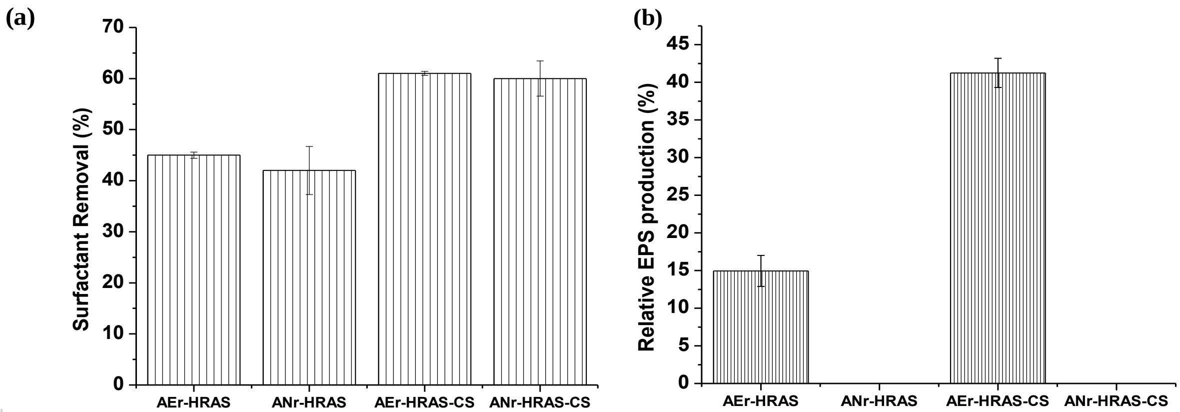

Figure 4.2: Results showing **a)** Percentage surfactant removal and **b)** relative EPS production under aerobic and sorption control anaerobic HRAS system configuration. **AEr**-Aerobic; **ANr**-Anaerobic; **CS**-Contactor Stabilization; **HRAS**-High rate activated sludge

The results obtained from EPS measurements (Figure 4.2b), showed a 63 % increase in relative EPS production for the optimized aerobic HRAS-CS system configuration compared to the conventional HRAS system. This result agrees with the results in section 4.6.1, that an increase in EPS production can potentially improve surfactant removal through enhanced biosorption. These results also suggests that surfactant removal was mostly through biosorption and not biodegradation. The EPS increase produced more sorption sites in the system, as also has been reported by Jimenez et al., 2015, Rahman et al., 2016 and Kinyua et al., 2017.

4.6.3 The impact of the HRAS-CS system configuration on OTE and alpha

The optimized HRAS-CS system configuration has been found to be more effective for surfactant removal compared to the conventional HRAS system. Several studies have also reported that surfactants are the major OTE suppressing agent and have a significant negative impact on OTE and alpha (Wagner et al, 1996; Hebrard et al, 2014 and Odize et a, 2017). However, the potential impact of the optimized HRAS-CS system configuration on OTE and alpha are yet to be established, even with its high surfactant removal capability. A series of pilot scale reactor tests as described in section 4.5.7 were performed to better understand the effect of the optimized HRAS-CS system on OTE and alpha. Results obtained from the pilot scale reactor test as shown in Figure 4.3a, showed no apparent change in the alpha of both the conventional HRAS and optimized HRAS-CS, even though high surfactant removal was observed for the latter (Figure 4.2a).

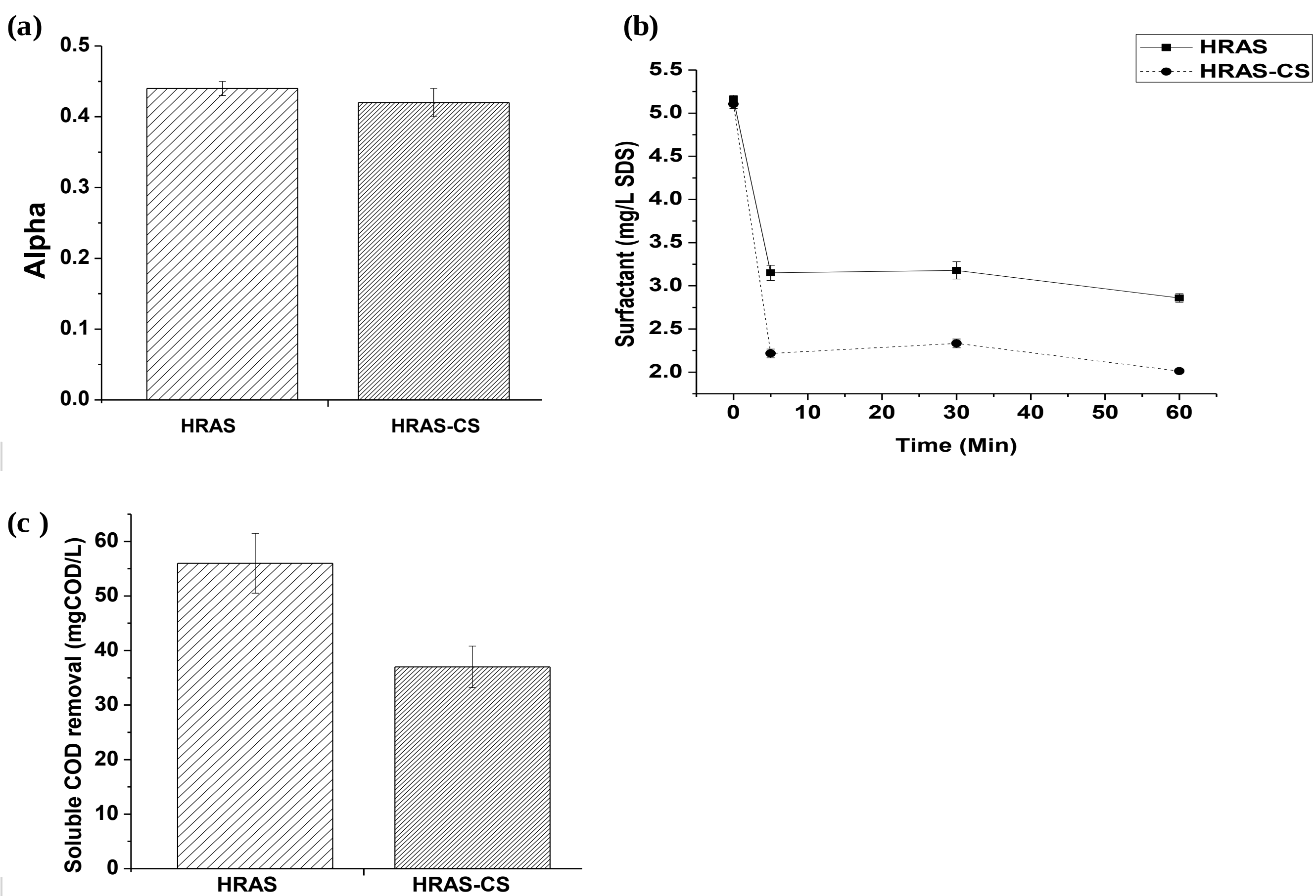

Figure 4.3: Results showing **a)** Alpha profile and **b)** Surfactant profile and **c)** Souluble COD removal in both conventional HRAS and optimized HRAS-CS system configuration.

The insignificant impact of the optimized HRAS-CS system configuration on alpha, even with a high surfactant removal was unexpected. Some studies have reported a negative influence of anionic surfactants adsorbed to microbial flocs in activated sludge which could lead to microbial cell degradation, activated sludge characteristics modification and enzyme inhibition (Cserháti et al., 2002; Liwarska-Bizukojc et al 2005 and Pang et al., 2006). All these could have played a role in negating the positive impacts of surfactant removal. As shown in Figure 4.3c, the conventional HRAS system had a higher soluble COD removal compared to the optimized HRAS-CS, which could have also led to surfactant oxidation and alpha improvement in the conventional HRAS system.

4.6.4 Evaluating different bioaugmentation ratios to determine the optimum bioaugmentation system configuration.

Wastewater bioaugmentation has been reported (Mahvi et al, 2004) to be an effective method of surfactant removal, which can also improve alpha. However, none of the studies show a direct correlation of surfactant removal with alpha. In addition, the relationship of the bioaugmentation ratio (HRAS: RAS (return activated nitrifying sludge)) with alpha has not been reported. To investigate this, a series of pilot scale reactor test as described in section 4.5.8, using different bioaugmentation ratios was conducted.

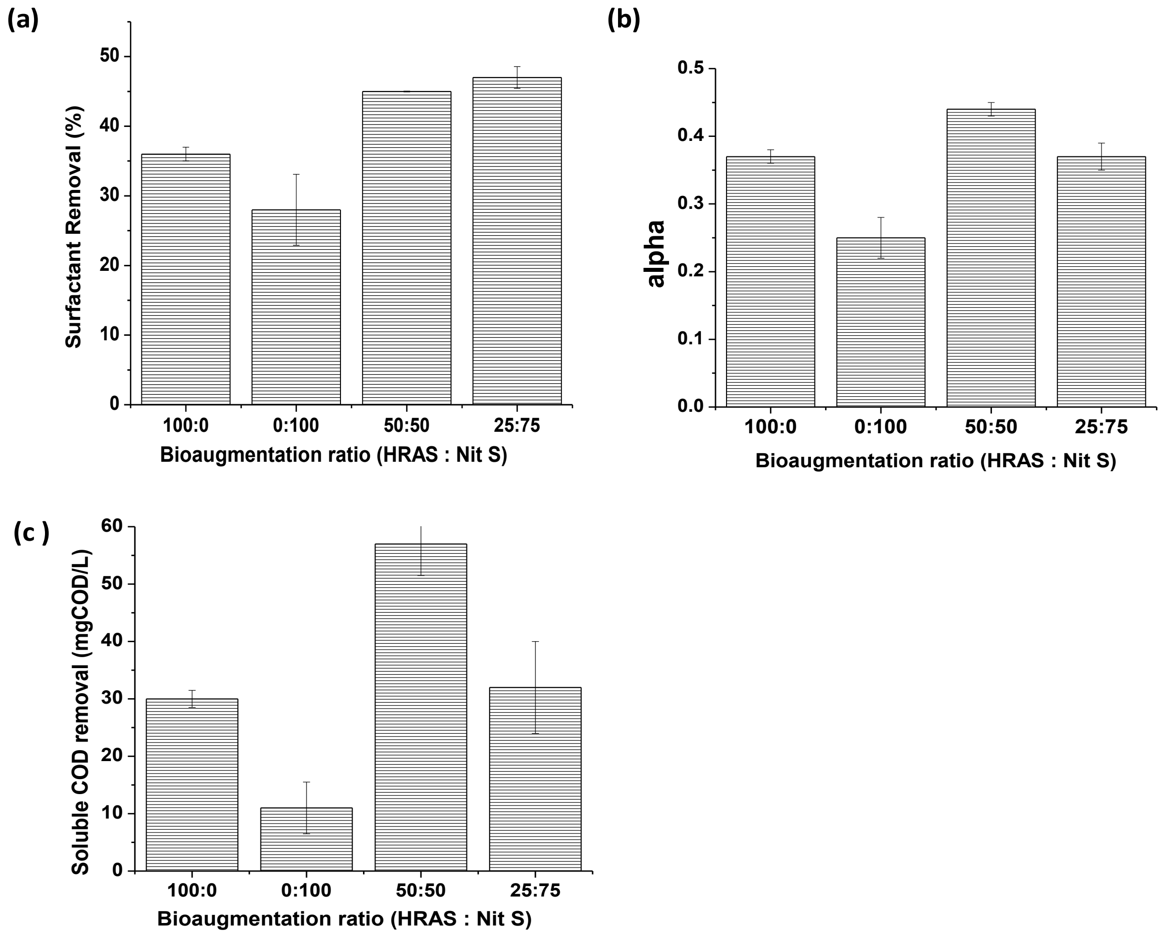

Figure 4.4: Results showing **a)** Alpha profile and **b)** Surfactant percentage removal and **c)** Souluble COD removal in different bioaugmentation ratio, conventional HRAS and Nitrification sludge **(Nit S)** configuration added to same CEPT effluent.

Figure 4.4a result shows that a 31 % surfactant reduction was achieved by the 50: 50 and 25: 75 (HRAS: Nit S) bioaugmentation ratio configuration compared to the non bioaugmented HRAS configuration. About 68 % more surfactant removal was achieved by both of the bioaugmented sludges (50:50 & 25:75) when compared to the non bioaugmented nitrifying sludge (0:100). However, alpha results shown in Figure 4.4b indicated an increase in alpha for only the 50:50 bioaugmented sludge (0.44 ±0.01) when compared to the non bioaugmented HRAS (0.37±0.02), Nit S (0.25±0.03) and 25:75 bioaugmented sludge (0.37±0.01). The low alpha obtained for the non bioaugmented Nit S and added CEPT effluent substrate, could be as a result of sorption of the surfactants onto the flocs whose presence in the reactor still has effect on the OTE and alpha. Under the non bioaugmented Nit S configuration, surfactant removal observed is primarily through sorption. That could explain why the high surfactant removal obtained in the 25:75 bioaugmented sludge did not show any impact on alpha, as more surfactant were sorbed compared to biodegradation. The high alpha shown in the optimum 50:50 bioaugmented sludge may be as a result of surfactant removal through the oxidation mechanism, complementing its removal through the sorption mechanism. As such, most of the sorbed surfactant to the microbial flocs are thought to be oxidized, thereby minimizing their effect on OTE and alpha. Soluble COD results from Figure 4.4c also showed a 78 % soluble COD removal for the 50:50 bioaugmented sludge configuration compared to the non bioaugmented HRAS, Nit S and 25:75 bioaugmented sludge. This result further supports the oxidation impact of the COD on surfactants, as more surfactants in the form of COD are removed during the COD oxidation process. The results in this study show that maximum surfactant removal and alpha increase are achieved at a bioaugmentation ratio of 50:50 (HRAS: Nit S). Mahvi et al., 2004 in their studies proposed that surfactant removal during wastewater treatment is achieved by sorption to the primary and secondary sludge and oxidation during the aerobic treatment.

4.6.5 Impact of an anaerobic selector (An-S) and its hydraulic retention time on surfactant removal OTE and alpha improvement.

To determine the impact of an anaerobic selector and selector hydraulic retention time on surfactants and alpha increase, several pilot scale tests were conducted as described in section 4.5.9. Results from Figure 4.5a (i) showed an increased surfactant removal when the simulated 50:50 bioaugmented sludge was modified, by introducing an anaerobic selector phase of 30 min as compared to the other reactor system configurations. The high surfactant removal also complemented the high alpha obtained in the anaerobic selector reactor as shown in Figure 4.5a (ii). About 135 % more surfactant was removed in the An-S reactor system when compared to the conventional HRAS system, while about 93 and 43 % more surfactant was removed when compared to the HRAS-CS and bioaugmented reactor system configuration, respectively.

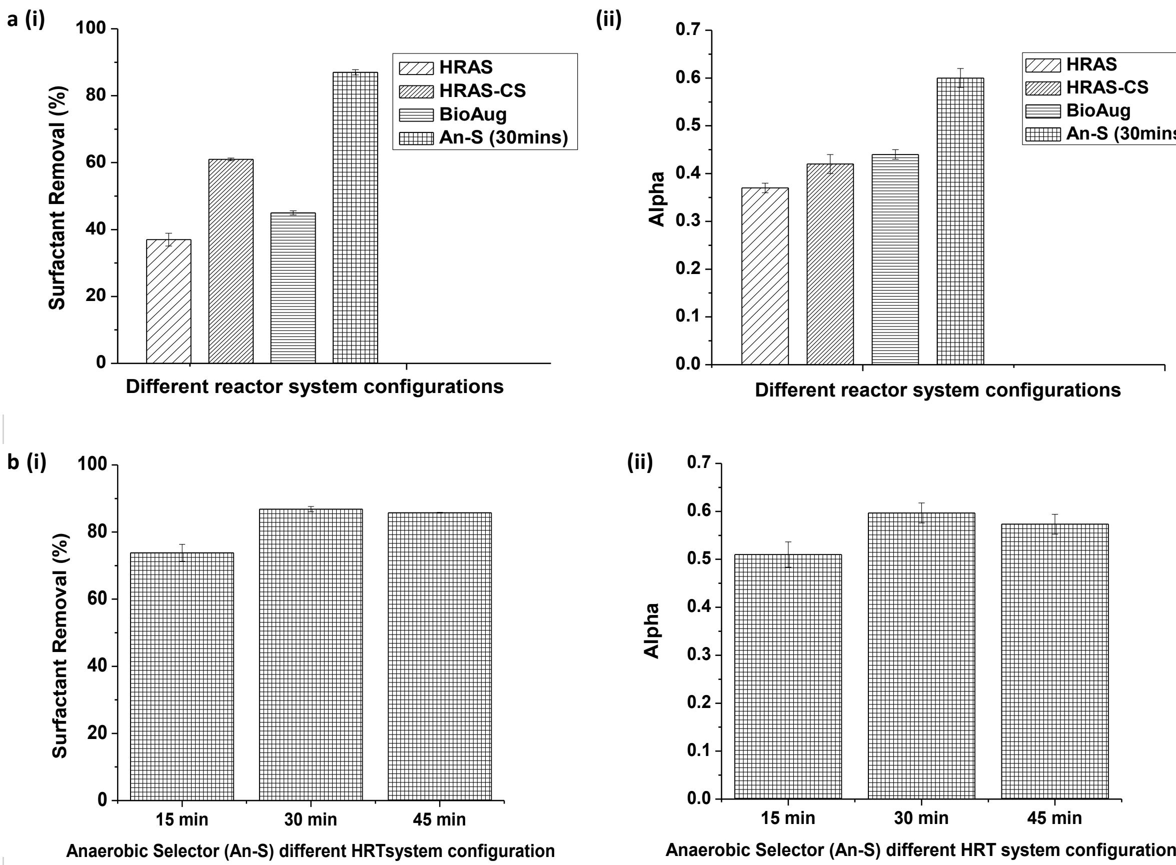

Figure 4.5: Results comparing **a)** different reactor system configuration studied **i)** surfactant percentage removal and **ii)** alpha profile; **b)** anaerobic selector **(An-S)** different HRT configuration **i)** surfactant percentage removal and **ii)** alpha profile.

According to some studies (Albertson, 2005; Azimi and Zamanzadeh, 2006), an anaerobic selector reduces soluble COD and enhances sorption during wastewater treatment. It also prevents filament proliferation and promotes development of biomass with increased sorption capacity and good settling characteristics. The high surfactant removal obtained in this study by the anaerobic selector configuration suggests an increased surfactant sorption capacity with an increasing biomass under the anaerobic conditions. However, the increased alpha could also be as a result of biodegradation or consumption of surfactants under anaerobic conditions. In the absence of oxygen under anaerobic conditions, denitrifying bacteria in the bioaugmented sludge which is nitrate rich, especially with the DC water sludge used in this study, may consume soluble COD as a carbon source and in the process and also biodegrades surfactants. Surfactants could be sorbed and also biodegraded. Lu et al., 2008 in their studies found that addition of nitrate or sulfate significantly enhanced anaerobic biodegradation of nonylphenols when they were added as an additional terminal electron acceptor. Another possibility that could have also played a role in the high surfactant removal is the intracellular storage of surfactants in microbial flocs under anaerobic condition, which could also be consumed as a carbon source by denitrifying bacteria.

Hydraulic retention time (HRT) is a very critical anaerobic selector design parameter, as bulking, filament growth and odor related complications can occur under long anaerobic HRT conditions. Albertson et al., 2005 reported that an anaerobic selector reduced soluble COD in 12 to 30 minutes of the retention time. Results shown in figure 4.5b (i) and 4.5b (ii) show maximum surfactant removal and increased alpha at 30 min HRT in the anaerobic selector, which correlates with past studies. DC Water, where this research was conducted, operates its anaerobic selector at about 45 mins, but this study showed that the highest surfactant removal and alpha increase was achieved at 30 min.

The introduction of the anaerobic selector to the already bioaugmented sludge suggest enhanced sorption and biodegradation of surfactants, which significantly increases alpha. More test are therefore recommended to explore the potential for modifying the anaerobic selector, bioaugmentation system.

4. CONCLUSION

a) High rate activated sludge contactor stabilization system configuration (HRAS-CS): Different stabilization times (3, 6 and 15 hrs) were evaluated for the HRAS-CS system technology, and the maximum surfactant removal and alpha was achieved at 3 hrs stabilization time.

b) Bioaugmented HRAS technology (Bioaug): HRAS was bioaugmented with nitrification/denitrification sludge at different ratios (100: 0, 0: 100, 50: 50 and 25: 50) and evaluated, and the maximum surfactant removal and alpha were observed at the 50: 50 bioaugmentation ratio.

c) Anaerobic selector phase technology (An-S): Different An-S technology HRT (15, 30 and 45 min) were also evaluated to determine the selector time for maximum surfactant removal and increased alpha. A 30 mins An-S HRT was observed to have the highest surfactant removal and increased alpha.

Each of the treatment technologies maximum surfactant removals were 37, 45, 61 and 87 %, and alphas of 0.37 ±0.01, 0.42 ±0.02, 0.44 ±0.01 and 0.60 ±0.02 for conventional HRAS, HRAS-CS, Bioaug and An-S reactor system configuration, respectively. The optimized bioaugmented anaerobic selector phase technology showed the highest increased surfactant removal (135 %) through enhanced surfactant biosorption and biodegradation. This study showed that the optimized bioaugmented anaerobic selector phase reactor system configuration is a promising technology or strategy for surfactant removal, which could help to minimize the impact of surfactants on alpha during secondary treatment.

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CHAPTER FIVE

CONCLUDING REMARKS AND ENGINEERING SIGNIFICANCE

This dissertation investigated key components of oxygen transfer in activated sludge systems ranging from diffuser fouling to the impact of wastewater characteristics on aeration performance. Strategies to alter the activated sludge processes to mitigate diffuser fouling and also enhance removal of "target pollutant(s)" that limit aeration performance were evaluated with a goal of reducing aeration costs. The benefits of this study are shown in three studies described in three separate chapters (2 to 4). Recommendations that will help in minimizing aeration energy costs associated with the activated sludge process are proposed at the end of this chapter.

Specific conclusions for each research objectives are as follows:

Objective 1: ***Evaluation of diffuser fouling dynamics, its impact on OTE and DWP, and the development a fouling mitigation techniques that can prevent or reduce fouling.***

Application of fine pore diffusers in wastewater treatments plants to save energy and operating costs has been limited by fouling, which reduces aeration efficiency and contributes to air blower pressure buildup. Moreover, there is insufficient information on the effectiveness of different types of fouling treatment techniques with respect to dynamic wet pressure (DWP) escalation and aeration efficiency. A study on fine bubble diffuser fouling dynamics and mitigation quantified by dynamic wet pressure (DWP) testing, oxygen transfer efficiency and alpha was carried out in a pilot reactor. DWP quantified the fouling dynamics of fine pore diffusers. A diffuser fouling physical treatment (reverse flexing, RF) method was able to mitigate fouling of the fine pore diffusers by preventing an increase in DWP normally observed in fouled fine pore diffusers. The RF treatment method reduced fouling by 35 % as compared to the control diffuser (without reverse flexing). This will reduce the pressure burden and air blower energy requirement. However, the RF treatment method did not show any significant impact in preventing a decrease in aeration efficiency (α F) of the treated fine pore diffusers. This is likely because the RF treatment method possibly maintained or increased pore size opening after flexing, causing the formation of large bubbles with reduced surface area and OTE. It could also have been the inability of the RF method to remove irreversible fouling, or a dominant impact of other wastewater constituent (i.e., surfactant which is a major oxygen transfer reducing contaminant) on oxygen transfer efficiency and alpha. This challenge led to the next research objective.

Objective 2: ***Determine the impact of different wastewater characteristics on aeration efficiency design parameters, OTE and alpha***

The effectiveness of oxygen transfer in the activated sludge wastewater treatment process is often defined by oxygen transfer efficiency and alpha, which was estimated in this study using a real-time self-calibrated off gas analyzer. Results from off gas testing performed on a series of pilot scale experiment using different wastewater treatment process effluents, showed a clear relationship between the wastewater treatment processes (raw wastewaters, primary effluent, HRAS effluent and nit/denit effluent) and alpha. Increased alpha values were observed for the cleanest wastewater effluents. The highly treated nitrification/denitrification effluent contained less surfactant and COD and had the highest alpha value. A 45 % increase in alpha was observed with a 92 % decrease in total COD and a 98 % decrease in surfactants. Results suggest a strong impact of surfactant and COD on OTE and alpha. Another series of pilot batch-scale experiments was performed to characterize the wastewater process pollutant (surfactant and COD fractions [soluble and particulates]) and their impact on OTE and alpha. Off gas test results showed that surfactant and particulate COD fractions were the constituents contained in wastewater that depressed OTE and alpha. Soluble COD did not show any inhibiting effect on OTE and alpha. The volumetric mass transfer coefficient (k_La) also showed that surfactants and particulate COD are negatively correlated, providing supporting evidence that surfactants and particulate COD in wastewater were responsible for oxygen transfer and alpha depression. The energy savings from effectively removing these oxygen transfer suppressants (.i.e., surfactants) prior to activated sludge treatment will help reduce energy costs. The next study objective was designed to evaluate different strategies of surfactant removal during activated sludge treatment.

Objective 3*: Evaluate three different optimized activated sludge treatment technologies to reduce surfactants through enhanced biosorption and biodegradation.*

Three optimized activated sludge wastewater treatment technologies were evaluated to determine their impact on surfactant removal through enhanced biosorption and biodegradation. These technologies were: 1) High rate activated sludge (HRAS) with contactor-stabilization technology (The contactor stabilization process) (HRAS-CS); 2) HRAS bioaugmented (BioAug) with nitrification sludge (Nit S); and 3) Bioaugmented HRAS with an anaerobic selector phase (An-S) configuration.

All three technologies increased surfactant removal through enhanced biosorption and biodegradation to various degrees when compared the conventional HRAS, but the *HRAS-CS* and *An-S* technology were much better than the BioAug treatment technology. The increased surfactant removal achieved by the HRAS-CS treatment technology was observed to be primarily through enhanced biosorption, and as such limited its ability to improving OTE and alpha, since the sorbed surfactants were still attached to the microbial flocs in the reactor. On the other hand, the *An-S* treatment technology achieved increased surfactant removal through both enhanced sorption and biodegradation, which could explain why a larger increase in alpha was achieved through this technology. The results suggest an enhanced biodegradation of sorbed surfactants under anaerobic condition because in the absence oxygen, denitrifying bacteria present in bioaugmented sludge consume soluble COD as a carbon source to produce nitrogen gas, and in the process also biodegrade surfactants. This study also established the optimum performance process conditions for each of the technologies.

Future research recommendations

Additional study of the *An-S* treatment technology is warranted. The optimized bioaugmented HRAS anaerobic selector phase reactor system configuration appears to be a promising technology or strategy to minimize the high energy consumption associated with secondary treatment.

More off gas measurements are also recommended to evaluate OTE and alpha using the modified anaerobic selector process, since results showed the potential ability of the technology to capture more surfactant through enhanced biosorption, although unmetabolized surfactants sorbed to the sludge matrix can negatively influence OTE and alpha. These research insights can provide operational guidelines for achieving better effluent quality and reducing the aeration energy costs. However, the results in this study were obtained using DC Water processes and optimization conditions at other plants might be different.

APPENDIX

A: SUPPLEMENTAL DATA FOR CHAPTER TWO

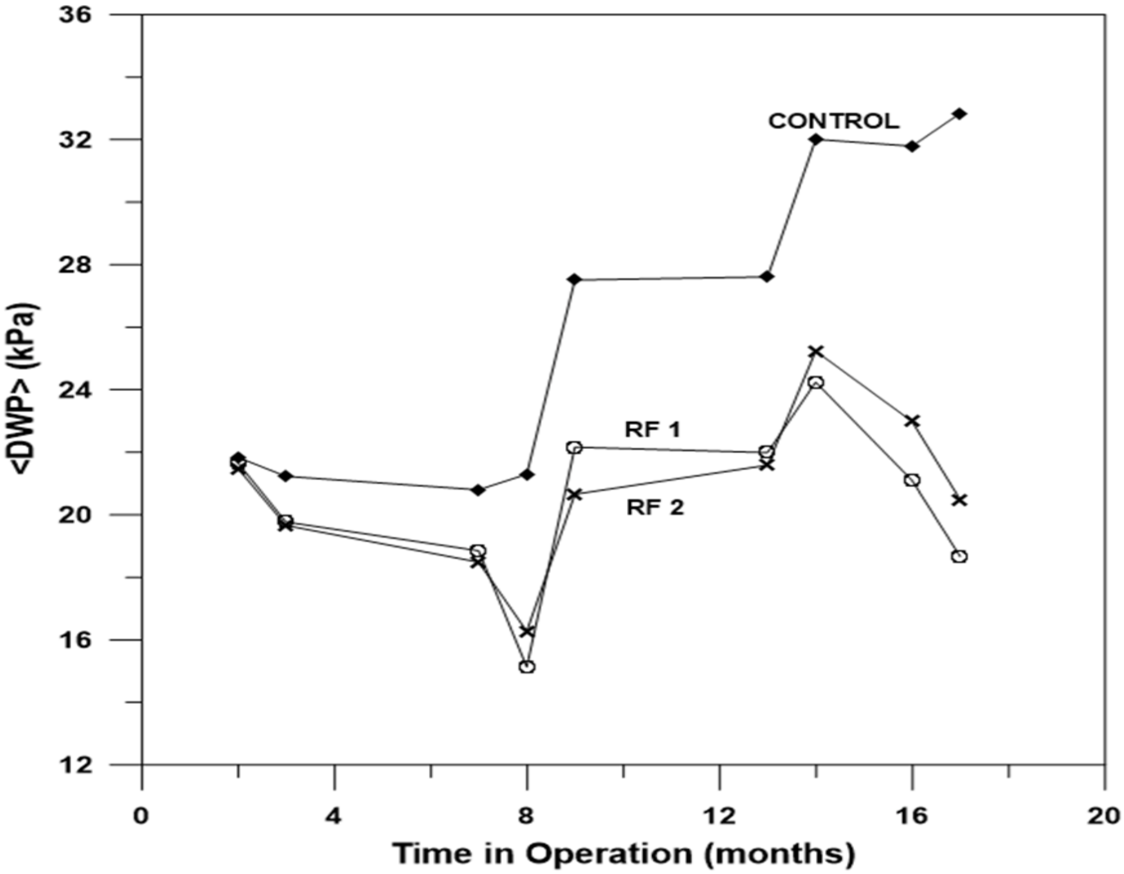


Figure S1. Impact of Reverse Flexing (RF) on weighted dynamic wet pressure (DWP) of polyurethane messner panel diffusers over time in operation

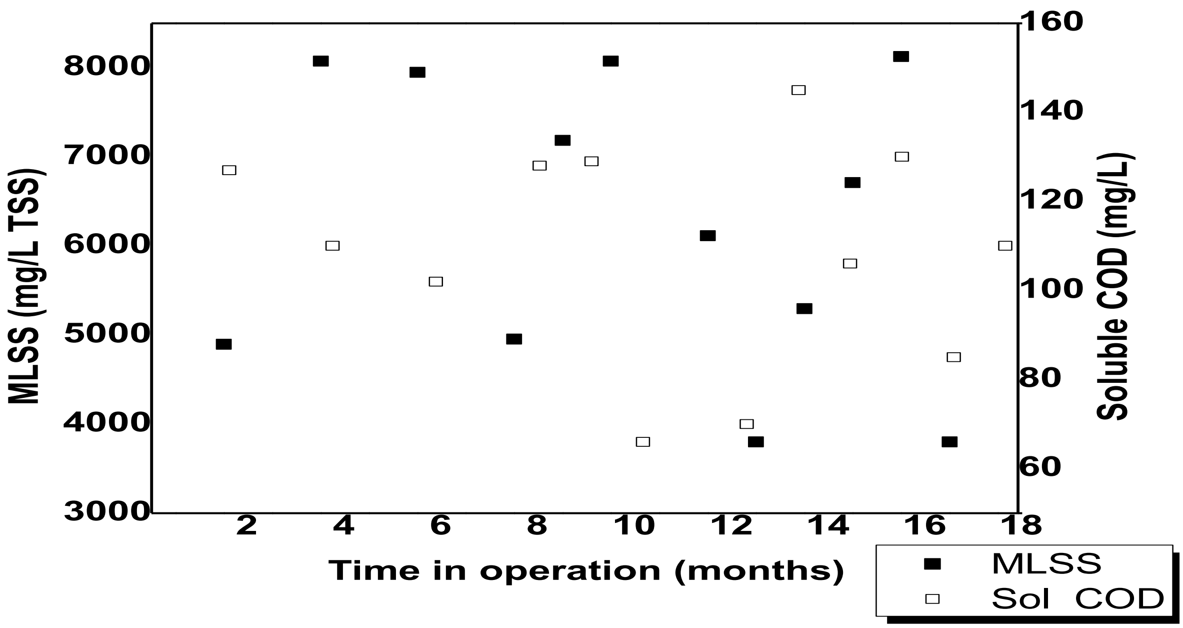


Figure S2: Measured MLSS and soluble COD profile over 17 months of operation

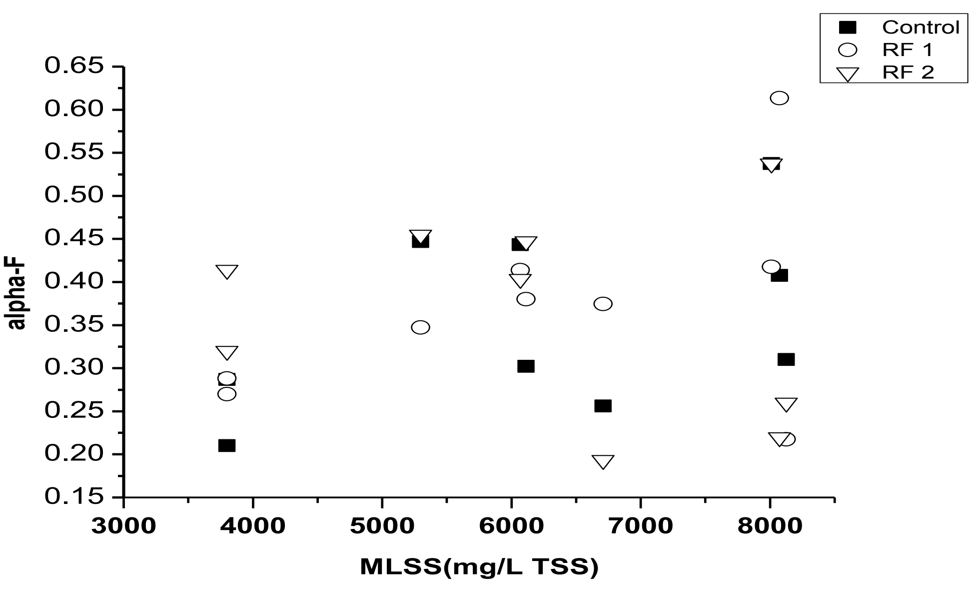


Figure S3: Comparing evaluated alpha fouling over measured MLSS over time in operation

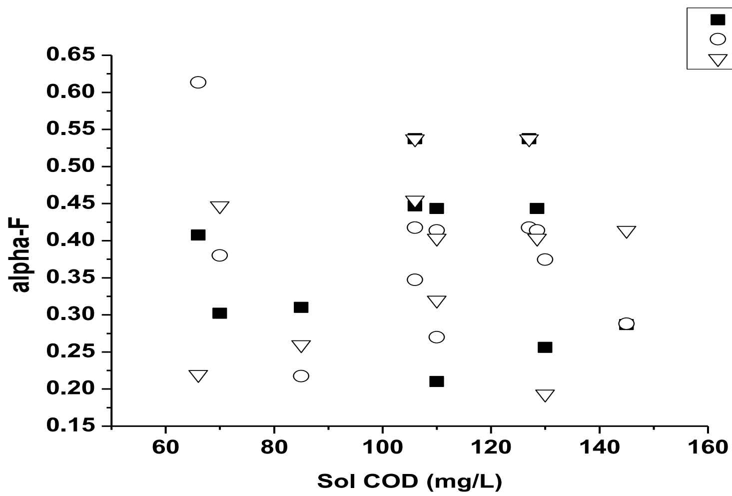
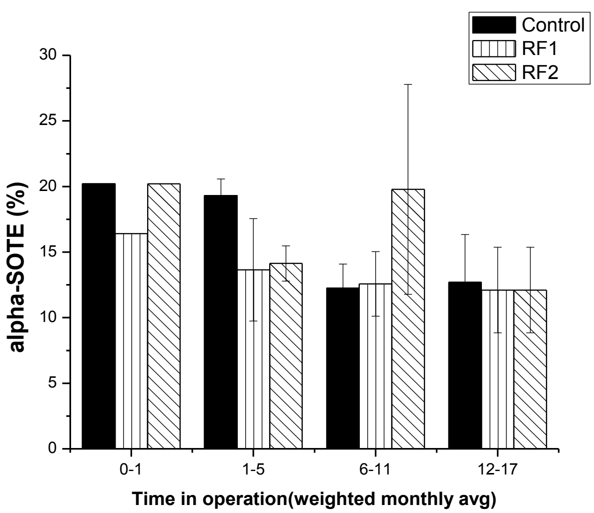


Figure S4: Comparing evaluated alpha fouling over measured soluble COD over time in operation



B: SUPPLEMENTAL DATA FOR CHAPTER THREE

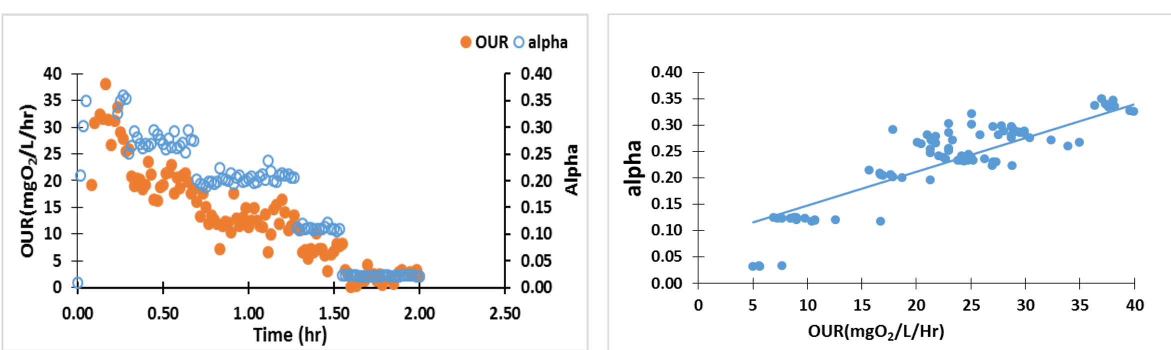
B.1 : **P values**

HRAS +	**Raw/CEPT effl**	**CEPT/Sec Effl**	**Sec /Nit effl**
	0.465473609	*1.31464E-05*	*7.92323E-05*

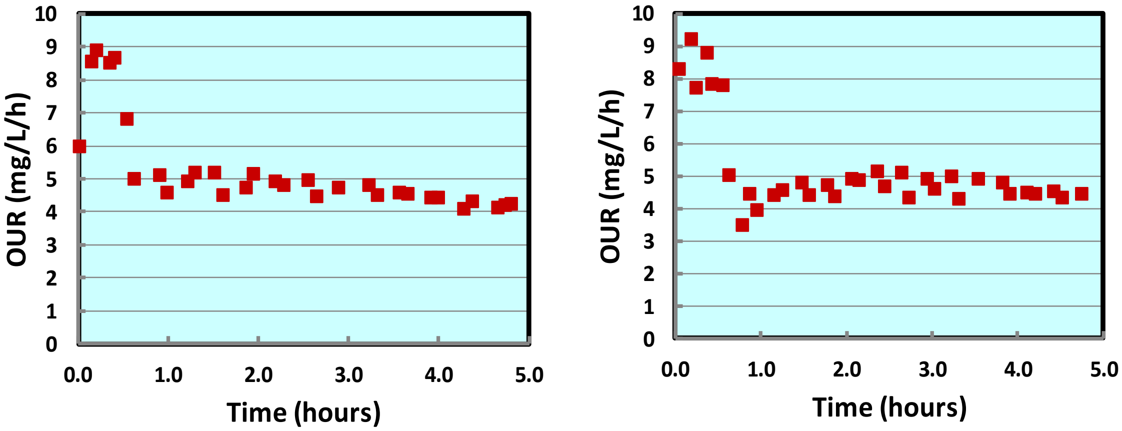
Surfactant conc.	**0/1mg/l**	**1/5mg/l**	**5/10mg/l**	**1/10mg/l**
	5.99337E-06	*0.219169*	*0.440298*	*0.034332*

Surfactant conc.	**0/1mg/l**	**1/5mg/l**	**5/10mg/l**
	2.39642E-05	*2.27529E-07*	*0.035458*

B. 2



Graph showing the alpha and OUR profile of HRAS + wastwater matrices in the 2 hrs pilot batch-scale experiment



Measured respirometric OUR curve of HRAS and wastewater matrices mixtures