SURFACE AERATION TO REDUCE ODOR GENERATION POTENTIAL IN SWINE MANURE LAGOONS

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Dedication

This dissertation is dedicated to my parents: Weizhu Cai and Bangrui Dong.

Abstract

The odors originating from open liquid manure storages are one of the most persistent problems and present undesirable environmental situations that have caused conflicts between swine producers and their neighboring communities. A considerable amount of research effort has been spent on controlling odor emissions from swine manure storages with methods studied including aeration and anaerobic treatment. Surface aeration is the most cost-effective method. Therefore, more information is needed about the system improvement and optimal depth determination of surface aeration.

This thesis presents investigations on gas injection in water, including a field-and-lab-scale study on surface aeration to reduce odor generation potential and laboratory-scale studies on aerator module development, liquid flow rate, alpha factor, surface aeration depth and temperature profiles on aeration efficiency. The purpose of the first investigation was to evaluate the effectiveness of a surface aeration system using venturi injectors, in which air was injected into a pipeline and discharged into the lagoon as a gas-liquid diffuser. This technique was shown to be an effective means of improving the dissolved oxygen levels and reduce odor generation potential in the lagoon. The purpose of the laboratory investigation was to better understand the effect of liquid flow rate, aerator module design, surface aeration depth and temperature on the surface aeration efficiency. Important fundamental results obtained in this thesis such as the field-scale surface aeration study and optimal surface aeration depth were very useful for many engineering applications, and pave the way for all animal producers (not just swine) who use liquid manure storages to adopt advanced aeration technologies for controlling manure odors.

Keywords: Aerator module development; Alpha factor; Diffuser submergence; Liquid flow rate; Odor reduction; Surface aeration; Temperature profile.

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List of Abbreviations and Definitions

$(k_L a)_{20}$ The overall mass transfer coefficient at T=20°C

Apha factor Ratio of the oxygen transfer coefficients in process to clean water at equivalent conditions of temperature, mixing, geometry

BOD Biological oxygen demand

C_∞ Determination point value of the steady DO concentration at time approaches infinity

$C_{\infty 20}$* The determination point value of steady-state DO concentration corrected to 20 °C and a standard barometric pressure of 1 atmosphere

COD Chemical oxygen demand

DO Dissolved oxygen

$k_L a$ Oxygen transfer coefficient in clean water

$k_L a\ _p$ The volumetric oxygen transfer coefficient in process water

ORP Oxidation-reduction potential

R^2 The coefficient of determination

SDS Sodium layryl sulphate

SOE Standard oxygenation efficiency

SOTR Standard oxygen transfer rate

SS Suspended solids

TS Total solids

TVS Total volatile solids

VFAs Volatile fatty acids

VS Volatile solids

Chapter 1 Introduction

1.1 Background and significance

At the beginning of 2009, the U.S. inventory of hogs and pigs stood at nearly 67 million head compared with 2.6 million in 1990 (Hunt and Vanotti, 2001; USDA, 2009). Associated with this increase is the air pollution problem (odors) which has become a center of public concern. This is reflected in the increased frequency of odor-related complaints in areas where swine production facilities are more intensified. Odor management is currently impacting many aspects of the swine industry and there appears a potential that the sustainability, productivity, and profitability of swine producers will be dependent upon whether they can reduce the emission of offensive odorants from operating swine production units to a level which surrounding communities could tolerate. Therefore, minimizing livestock manure's impact on the environment is one of the major challenges facing U.S. agriculture (Vanotti et al. 2007), with identified environmental and health concerns including emissions of ammonia (Aneja et al., 2000; Szogi et al., 2005), odors (Schiffman et al., 2001; Loughrin et al., 2006), pathogens (Sobsey et al., 2001; Vanotti et al., 2005a), and water quality deterioration (Mallin, 2000). Among these issues, the odors originating from open liquid manure storages are one of the most persistent problems and present undesirable environmental situations that have caused conflicts between swine producers and their neighboring communities (Donham, 2000; Iverson et al., 2000; Schenker et al., 1991; Schenker et al., 1998). Therefore, there has always been an unfading interest in developing swine manure treatment systems that can address the odor problem resulting from open liquid manure storages.

A considerable amount of research effort has been spent on controlling odor emissions from swine manure storages with methods studied including: management of waste loading (Lim et al., 2003), use of additives (McCrory and Hobbs, 2001), mechanical aeration (Heber et al., 2002; Westerman and Zhang, 1997), and use of covers (Clanton et al., 1999; Funk et al., 2004; Miner et al., 2003; Zahn et al., 2001b). Among these methods, aeration has been extensively researched for the control or reduction of lagoon odors (Williams et al., 1984; Williams et al., 1989; Pain et al., 1990; Sneath et al., 1992; Zhang et al., 2004; Zhang and Zhu, 2005). In these studies, often the mechanical aerators that are used in aeration of other wastewaters are applied to liquid swine manure storages. Cumby (1987) summarized the mechanical aerator types tested in water and wastewater aeration including compressed air, mechanical surface, mechanical subsurface, and combined compressed air and venturi. However, his study has not led to a wide adoption of aeration in controlling lagoon odors at field scale due to a number of obstacles. A major one may lie in the prohibitive operating cost to aerate the entire lagoon content.

To address the cost issue, many researchers have turned their attention to surface aeration with much work done at lab scale over the years by stratifying anaerobic lagoons (Humenik et al., 1980; Schulz and Barnes, 1990; Zhang et al., 1997; Zhu et al., 1999; Zhang et al., 2000; Zhang and Zhu, 2003, 2005). A stratified lagoon is created by introducing mechanical aeration into the top layer while maintaining anaerobic conditions in the lower layers. To date, the depth of the aerated surface layer ranging from 80 to 600 mm has been studied (Humenik et al., 1980; Ginnivan, 1983; Zhang et al., 1997; Zhu, 2001; Zhang et al., 2004; Zhang and Zhu, 2005). The phenomenon of stratification, peculiar to deep lagoons, plays a fundamental role in the function of the surface aeration system (Kellner and Plres, 2002) because the aeration depth affects the vertical mixing and re-stratification of the lagoon, which then influences the water chemistry and microbial processes in different layers (Gu and Stefan, 1995). Therefore, there should exist an optimal depth of the surface aerated layer that dissolved oxygen can be replenished in that layer while still preserving stratification. Unfortunately, little literature information is available about the optimal depth of surface aeration for swine manure storage to influence water chemistry and biological processes in both the aerobic and anaerobic layers. To that end, research on determining the optimum depth of surface aeration is needed that is critical for the effective and efficient use of surface aeration technology in swine manure treatment for odor control.

The success in determining the optimum depth of the aerated layer, however, may not guarantee a smooth sail in technology transfer. It is not in dispute that the major hindrance of using aeration to control odor rests with the fact that most commercial aeration systems are expensive relative to production costs: for a better effluent with low organic strength, the treatment costs tend to go higher. Conversely, the low-cost options tend to equate with less, and sometimes inadequate, treatment. This constitutes the reason why the tremendous work done by the past researchers has all fallen short of transferring this technique into field-scale operation. To overcome this problem and advance the knowledge for field application, there is certainly an acute need to develop better aerators that can increase aeration efficiency without increasing energy consumption. Only in this way can aeration become a truly cost-effective method to control odor.

In order to evaluate the effectiveness of an odor control technology, a measurable parameter(s) has to be determined. Unfortunately, consensus has not been reached on odor indicators among researchers working in this area. Evaluation of odor-abating waste handling methods consists of sensory analysis by olfactometry relying on human subjects or analytical methods. The olfactometry approach is cumbersome for routine analysis because rather large odor panels are needed to obtain reliable and reproducible results (Gralapp et al., 2001). For odor evaluation, a simplified approach was used by measuring the concentration of selected odorous compounds using water quality parameters as indicators in liquid manure (Loughrin et al., 2006). Since biological oxygen demand (BOD) is a measure of the total biodegradable organic materials (including all odorous compounds), its use as an indicator of the odorous compounds levels has been suggested and used by a number of researchers in evaluating liquid swine manure treatment systems for odor control (Williams, 1984; Thacker and Evans, 1985; Evans et al., 1986; Burton, 1992; Loughrin et al., 2006). Besides, the use of volatile fatty acids (VFAs) to gauge odor offensiveness of swine manure has also become accepted by many researchers (Evans et al., 1986; Sneath, 1988; Sneath et al., 1990; Schulz and Barnes, 1990; Miller and Varel, 2001; Sheridan et al., 2003). Therefore, in order to get parallel information about the effectiveness of surface aeration on odor control, both BOD and VFAs will be used as odor indicators in this study.

The significance of the proposed work can be summarized in the following aspects: 1) it will advance the knowledge and understanding of the reaction kinetics and nutrients transport in the aerated surface layer in a swine manure storage; 2) it will enhance the engineering practice in aeration system design by providing key information with which the efficiency and effectiveness of the treatment can be greatly improved and/or optimized (i.e. developed aerator module, optimal surface aeration depth); 3) it will provide applicable knowledge to swine producers in selecting surface aeration equipment that meets their respective odor control needs; and 4) it will pave the way for all animal producers (not just swine) who use liquid manure storages to adopt advanced aeration technologies for controlling manure odors.

1.2 Objectives

The major objective of this study is to use surface aeration system to reduce odor (two indicators: BOD and VFAs) at lab and field scales. Aerator module development using clean water test and the optimal surface aeration depth determination are also conducted in this research.

The specific objectives of this research are:

(1) Conduct experiments to collect data in an aerated storage at lab and field scales to evaluate the liquid manure parameters and odor reduction potential of the experimented surface aeration system and the relationship between two odor generation indicators: BOD and VFAs.

(2) Present particular information on the design and evaluation of aerator modules composed of venture air injectors in terms of oxygen transfer efficiency, in which one, two, and three venturi air injectors were connected in two different configurations, either in series or in parallel, to determine the best aeration efficiency and oxygenation capacity.

(3) Gain a better understanding of some important factors (such as aeration depth, total solids content, and temperature) that may impact the effectiveness of the surface aeration method.

(4) Study on temperature profiles and aeration efficiency at varied diffuser depths and then determine an optimal surface aeration depth.

Chapter 2 Literature review

2.1 Review on odor control technologies

There are two basic principles for controlling odors from open liquid manure storages: (1) reduction of odors at the generation sources, and (2) removal of odors from collected gaseous streams before the odors are discharged into the atmosphere (Rappert et al., 2005). Source control is always the first choice for odor control.

A range of odor management techniques for source control are available to producers including: dietary manipulation (Hobbs et al., 1996, 1997); biofilters (Martin et al., 1996; Sun et al., 2000); covers for storage and treatment structures (Meyer and Converse, 1982; Sommer et al., 1993; Hodgson and Paspaliaris, 1996; Zhang and Gaakeer, 1998; Xue et al., 1999; Hornig et al., 1999; Clanton et al., 1999; Hudson et al., 2006); improved digester techniques (aerobic and anaerobic), with or without improved solids separation devices (Sneath et al., 1992; Tao et al., 1998; Beline et al., 1999; Paing et al., 2001); composting of biosolids (Vuorinen and Saharinen, 1997; Tiquia et al., 1998; Hong et al., 1998; Jeong and Kim, 2001), and advanced treatment techniques, such as activated sludge processes, upflow sludge bed digesters, and sequencing batch reactors, with or without off-gas treatment facilities (Subramaniam et al., 1994; Sanchez et al., 1995; Powers et al., 1997; Dugba and Zhang, 1999). This review will mainly focus on the available literature information related to covers, anaerobic treatment and aeration technologies.

2.1.1 Lagoon covers

An approach to diluting lagoon emissions is to contain the gases by using various types of covers over the entire lagoon surface. Floating synthetic covers have been used for lagoons and other liquid storages for many years (Sagley and Westerman, 1989; VanderZaag, 2008). These impermeable floating covers collect the gaseous products of manure decomposition but form raised bubbles that are vulnerable to high winds (Funk et al., 2004). Floating permeable biomass covers, such as barley straw or chopped cornstalks, are partially effective in reducing odors; however, the covers must be repaired or replaced frequently, involving recurring expenses (Clanton et al., 1999).

2.1.2 Anaerobic treatment lagoons

Anaerobic lagoons are a process in which microorganisms are used under anaerobic conditions to convert biodegradable organic materials to odorless gases, such as methane and carbon dioxide, and non-biodegradable solids. There are basically three steps involved in the process, i.e., hydrolysis, acidogenesis, and methanogenesis. The key to preventing odor production is that the balance between the second and third step has to be maintained. In other words, the production of acids by the indigenous bacteria and the consumption of acids by the methanogens to produce methane and carbon dioxide have to be in equilibrium. Otherwise, malodor may result. These processes are more sensitive to changes in temperature and loading rate as compared to aerobic processes. Influent wastewater flow is usually near the bottom of the lagoon and has a pH between 7.0 and 8.5. It is not mechanically mixed, although some gas mixing can occur. A scum usually develops at the surface, reducing heat loss and thus ensuring anaerobic conditions. Anaerobic lagoons contain bacteria that decompose organic matter to gaseous end products in two major steps. First, facultative microorganisms degrade carbohydrates, proteins, and fats to organic acids during an acid fermentation phase. Second, methane-producing microorganisms break down organic acids to produce methane, carbon dioxide, and other volatile products (Mulligan and Hesler, 1972). As a result, concentrations of volatile solids (VS) and carbon decrease over time (Oleszkiewicz and Koziarski, 1986). Anaerobic lagoons can handle a wide variety of waste characteristics. Typical reductions of up to 97% BOD, 95% SS and 96% COD are reported by US-EPA (2002).

Anaerobic lagoons have been used with many waste management systems throughout the United States to provide practical treatment and storage of swine manure (Westerman et al., 1990). The attractive features of anaerobic lagoons used in swine waste management systems include relatively low capital and operating cost, low labor and energy requirement, simplicity, and ease of management (Fulhage, 1980). Unfortunately, many anaerobic lagoons do not function as properly as designed due to overloading and bad management and complaints about the odor generated from these lagoons have risen widely. Even oversized lagoons (two to four times larger than standard design) can produce objectionable odor (Sweeten, 1980). In addition, anaerobic lagoons can become more odorous when they are overloaded due to: 1) sludge buildups that reduce treatment volume, 2) additional loading from herd expansions, and 3) cold weather that slows biological activity. Since organic matter is incompletely digested during winter, bacteria have excess organic matter to stabilize during warmer temperatures in spring. Increased biological activity coupled with thermal instability causes vigorous activity to occur along with burst releases of odorous compounds (Hamilton and Cumba, 2000). Lagoon odors therefore are typically worse in spring compared with other seasons of the year. Lower loading rates, especially during winter and early spring, can reduce odor release (Safley et al., 1993). Maintaining constant water levels and adding wastes frequently rather than in large irregular quantities also reduces odor release. Properly designed and operated anaerobic lagoons will usually be less odorous than overloaded and poorly managed lagoons (Ritter, 1989).

It could be inferred that the capability of treating swine manure by anaerobic lagoons could be limited due to the low temperatures, and more critical management appears needed to avoid overloading. As a matter of fact, overloading has become a major problem for the anaerobic lagoon systems currently used in the middle and southern areas in the United States due to the fast-growing swine industry producing a huge volume of manure in a short time period. Therefore, the balance between the acids produced by the indigenous bacterial groups and the acids consumed by the methanogens for methane formation can hardly be achieved. This is why there is little methane formed in, and there is strong offensive odor generated from, the waste storage lagoons.

2.1.3 Aeration treatment

Attempts to employ aeration treatment have been made to reduce offensive odors of swine manure for many years. Successful aeration systems for treating swine manure at the laboratory (Zhang et al., 1997), pilot (Yang and Gan, 1998), and full or farm scale

(Westerman and Bicudo, 2002; Yang and Wang, 1999; Westerman, 2005) have been reported. Most of the studies on aeration lagoons have shown success for odor control (Zhang et al., 1997; Schulz and Barnes, 1990; Ginnivan, 1983).

Aeration is a technical version of the biological oxidation of organic substances in liquid manure and represents a modern process of the liquid manure treatment. Under aerobic conditions, biodegradable organic materials in swine manure can be oxidized into stable inorganic end products by aerobic bacteria (Westerman and Zhang, 1997). A complete oxidation process can be expressed as:

$$\text{Organics (C, H, O, N, S)} + O_2 \longrightarrow CO_2 + H_2O + NO_3^- + SO_4^{2-}$$

Under aerobic conditions, the nitrogen compounds (proteins, peptides, and amino acids) are converted to ammonium (NH_4^+) by heterotrophic bacteria (require nourishment from organic substances) and then oxidized by autotrophic bacteria (obtain nourishment from inorganic matter such as NH_4^+ to nitrite (NO_2^-) and then to nitrate (NO_3^-)). Sulfur compounds, i.e., sulfur-containing protein, in the wastes are converted to sulfate (SO_4^{2-}) in the aerobic environment instead of odor-causing sulfide and mercaptan compounds in the anaerobic environment. The degree of oxidation depends on the amount of oxygen provided and the reaction time allowed in the treatment process. When the aeration is terminated and the dissolved oxygen is depleted, the environment is considered to be anoxic. Under these conditions, nitrate and sulfate function as oxygen for aerobic bacteria, and are reduced to nitrogen gas (N_2) and hydrogen sulfide (H_2S), respectively.

In addition, aerobic treatment of manure does not allow accumulation of volatile fatty acids and various other intermediate odorous compounds during treatment and this reduces odor problems during treatment processes. Once the stabilization has occurred, the manure can invariably be stored for longer periods without odor problems (Ndegwa, 2003). It is well established that during aeration treatments, the oxygen transfer rate and efficiency is affected by factors such as: the surface area in contact with air, mixing/turbulence, temperature, and the amount of solids or other constituents in the manure.

There have been reports indicative of promising results in using aeration for odor control (Williams et al., 1984; Williams et al., 1989; Pain et al., 1990; Sneath et al., 1992; Zhang et al., 2004; Zhang and Zhu, 2005). These techniques include continuous aeration (Burton and Sneath, 1995), intermittent aeration (Zhang et al., 1997; Zhu, 2001), combined anaerobic-aerobic bioreactors or mixed and aerated ponds (Bernet et al., 2000; Westerman and Bicudo, 2002), aerated lagoon bioblanket (Zhu, 1998), and surface aeration or shallow aeration (Ginnivan 1983 a,b; Zhang et al., 1997).

Among these techniques, surface aeration may be the least costly but has the potential to become a means for effective odor control for open manure storage. A stratified lagoon is created by introducing mechanical aeration into the top layer while maintaining anaerobic conditions in the lower layers. The resulting diphasic lagoon has the benefit of anaerobic sludge decomposition at the bottom of the lagoon, and positive odor control and scum elimination at the surface (Heber et al., 2002). Research in North Carolina (Humenik et al., 1975, 1980; Barker et al., 1980) indicated that odor control could be effectively achieved using aeration of the surface layer (0.60 m) of a deep lagoon (3.00 m or deeper). Ginnivan (1983) concluded from surface aeration studies in his laboratory that shallow surface aeration to depths ranging from 0.08 to 0.40 m was effective for odor control. Unfortunately, there is little information available about pilot-scale surface aeration systems on odor control from actual lagoons.

Zhang et al. (1996) conducted laboratory research to determine the aeration requirement for odor control of anaerobic lagoons by surface aeration. Four levels of DO concentrations in the top liquid layer (0.50, 1.50, 2.50, and 5.00 mg/L), three depths of aeration (0.15, 0.30, and 0.45 m), and two levels of aeration frequency (continuous and intermittent) were studied using 1.83 m tall columns. The columns were operated to simulate the anaerobic lagoon environments with swine manure. The results showed that continuous surface aeration at a depth 0.15 to 0.30 m with approximately 0.50 mg/L DO effectively controlled odors. Zhang and Zhu (2003) found that the BOD removal efficiency increased linearly from about 7.5% to 90% in the first four weeks and remained stable around 90% to 95% after first four weeks of surface aeration.

All these previous results indicate that surface aeration can be an effective tool in reducing odors from open manure storage structures. However, the field-scale study and the relationship between the aeration depth and the odor (and/or BOD, VFAs) reduction has not been studied sufficiently. And little is known as to how to improve, and predict, the treatment efficiency with different surface aeration schemes by optimizing the aeration depth.

2.2 Review on use of venturi aeration technology

The aeration equipment commonly employed in the wastewater treatment field consists of air diffusion units; turbine aeration systems in which air is released below the rotating blades of a submerged impeller; static aerators where air bubbles are released under submerged vertically mounted cylindrical tubes that contain baffles; and surface aeration units in which oxygen transfer is accomplished by high surface turbulence and liquid spray (Kim et al., 2005). Bloxham (1996) tested four types of aerator (venturi aerator, perforated small-bore aerator, surface aerator and diffuser) under summer and winter conditions on a full-scale pilot plant developed on a pig farm with respect to their efficiencies and operating systems and reported that venturi aerators were very reliable, no breakdowns, no maintenance, and effective for odor control.

Venturi aeration has become popular in recent years. The venturi (Figure 2.1) is a device which has been used over many years for measuring the discharge along a water pipe in water/hydraulic engineering. Air vacuum processes with venturi, which are simple in design and easy to operate, have been used in a wide variety of industrial and water purification processes (for example, ozone and chlorine injecting/contacting for drinking water, chemical injection such as liquid fertilizers in sprinkler irrigation system and aeration/dissolved oxygen transfer in drinking and waste water treatment). Many researchers studied the use of venturi tubes in water aeration systems (Baylar and Emiroglu, 2003; Emiroglu and Baylar, 2003; Baylar et al., 2005-2007; Bagature, 2005; Ozkan et al., 2006; Baylar and Ozkan, 2006; Zhang et al., 2003; Zhu et al., 2007; Gourich et al., 2008) and the results showed that venturi tubes had high air injection efficiencies and could be used as highly effective aerators in water aeration systems. Most recently, the detailed theory and advantages of using venturi tubes in water aeration systems were reported by Baylar and Ozkan (2007).

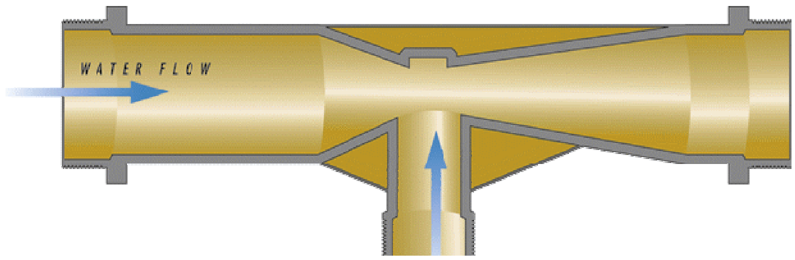


Figure 2. 1 Venturi injector (from Mazzei injector company website)

The fluid flowing in the pipe is fed through a contraction section to a throat, which has a smaller cross-sectional area than the pipe, so that the velocity of the fluid through the throat is higher than that in the pipe. This increase of velocity is accompanied by a fall in pressure. Beyond the throat the fluid is decelerated in a pipe of slowly diverging section, the pressure increasing as the velocity falls.

At the inlet of the venturi device, water flow has pressure energy. This pressure energy is converted to velocity in the converging cone. Water flow changes into a high-velocity jet stream. The increase in velocity through the venturi converging cone, caused by the differential pressure, results in a decrease in pressure. Thus, a vacuum is created at the suction port where water mixes with air. Two-phase flow (bubbly flow or air-water mixture) thus occurs. In the bubble flow, the absorbed gas phase (air) flows as discrete bubbles in liquid (water). This occurs when a minimal amount of differential pressure exists between the inlet and outlet regions of the venturi device. At the outlet of the venturi device, pressure energy is regained for mass transfer of air, which makes the device a viable, potentially economic, and competitive alternative for gas-liquid transfer. In mixing, no mechanical agitation is employed with the water pump being the only component to generate vigorous fluid motion (water and air, which are two phases). This device also has lower energy requirements, installation, operating and maintenance costs in tanks, ponds or lagoons. Based on these advantages, the venturi injectors were chosen to use as aerators in this study for a surface aeration system to reduce odor generation potential from open liquid swine manure storages.

2.3 Review on two odor generation potential indicators: BOD and VFAs

Odorous emissions from manure consist of a complex array of compounds such as volatile fatty acids, indoles and phenols, ammonia, volatile amines, and volatile S-containing compounds (Zhu, 2000). Subjectivity in the measurement and quantification of odor has always been a big hindrance in the development of strategies toward odor solutions. Evaluation of odor-abating waste handling methods consists of sensory analysis by olfactometry with human subjects or analytical methods. The olfactometry approach is cumbersome for routine analysis because rather large odor panels are needed to obtain reliable and reproducible results (Gralapp et al., 2001). And this method that uses the human nose is not only limited in their use by this subjectivity but also by the cost and difficulty of collecting representative samples (Hamilton, 1999).

Since aeration treatment enhances the decomposition of organic compounds in manure, evaluating the effectiveness of odor control by aeration treatment can therefore center on the understanding of dynamic change in VFAs, and BOD over the course of such a treatment. To that end, a simplified approach was proposed by measuring the concentrations of selected odorous compounds using water quality parameters as indicators in liquid manure (Loughrin et al., 2006). Since BOD is a measure of the total biodegradable organic materials (including all odorous compounds), its use as an indicator of the odorous compounds levels has been practiced by a number of researchers in evaluating liquid swine manure treatment systems for odor control (Williams, 1984; Thacker and Evans, 1985; Evans et al., 1986; Burton, 1992; Loughrin et al., 2006). Besides, the use of VFAs to gauge odor offensiveness of swine manure has also become accepted by many researchers (Evans et al., 1986; Sneath, 1988; Sneath et al., 1990; Schulz and Barnes, 1990; Miller and Varel, 2001; Sheridan et al., 2003). Therefore, in order to get more parallel information about the effectiveness of surface aeration on odor control, both BOD and VFAs were used in this study as odor indicators.

2.3.1 BOD

There have been reports on the relationship of BOD with the odor offensiveness of swine manure. Williams (1984) conducted a study to relate the chemical characteristics of liquid swine manure to its odor using human sniffing panelists and found that the offensiveness of the manure odor correlated linearly with the logarithm of BOD in aerobic systems. Similar results were also presented by Thacker and Evans (1985). Evans et al. (1986) found a log-linear relationship between odor offensiveness and the oxidation-reduction potential (ORP) in the aerobically treated swine manure and concluded that BOD was a suitable indicator for odor offensiveness because there existed a linear relationship between BOD and ORP. Burton (1992) did a review on aerobic treatment of swine manure and concluded that reducing BOD levels would stabilize the manure during storage; hence reducing offensive odors.

In another study, Loughrin et al. (2006) evaluated the use of certain water quality parameters as indicators of probable odorous compounds levels in wastewater systems. They presented data on the concentrations in liquid of six selected malodorous compounds (*phenol, p-cresol, p-ethylphenol, p-propylphenol, indole, and skatole*) and 15 water quality parameters measured at three successive stages of the treatment process and found that BOD was linearly related to total odorous compounds concentrations in the liquid of anaerobic and aerobic swine lagoons. And the correlation coefficient between BOD and the total odorous compounds was 0.97. Based on all these studies, BOD is chosen as an indicator of odor generation potential in the simulation model of surface aeration in this study.

2.3.2 VFAs

Typical acids in this group consist of *acetic*, *propionic*, *butyric*, *iso-butyric*, *valeric*, *iso-valeric*, *caproic*, and *capric acids* (Zhu, 2000). The VFAs can be produced from the determination of amino acids that are produced during the process of protein degradation and breakdown of carbohydrates. Williams (1984) first reported that VFAs could be a suitable indicator of manure odors. Since then, the use of VFAs to gauge odor intensity of swine manure has become accepted by many other researchers (Evans et al., 1986; Sneath, 1988; Sneath et al., 1990; Schulz and Barnes, 1990; Miller and

Varel, 2001; Sheridan et al., 2003). In addition, the VFA concentrations and odor threshold/odor quality scales were numerously employed in evaluating the effectiveness of techniques for swine manure odor reduction (Zhang et al., 1997; Zhu et al., 1999; Zhang et al., 2000). Research showed that VFAs were mainly formed from the solids in the smallest particle size range (<0.075 mm) in manure by microbial degradation (Yasuhara et al., 1984; Zhu et al., 2001).

2.3.3 Relationship between BOD and VFAs

The relationship between BOD and VFAs was studied and reported by many researchers. Zhu et al. (2001) found that the BOD concentrations correlated well with the VFA levels in the manure. Bell (1970) also found a close relationship between volatile fatty acids and odor offensiveness of anaerobically and aerobically stored poultry manure. A review paper by Spoelstra's (1980) indicates that for manure slurry stored anaerobically, VFAs, indoles and phenols are suitable indicators of odors. Another study by Williams (1984) found good correlations between odor offensiveness and both VFAs and BOD. Williams (1984) further established that not only were better correlations obtained with supernatant's VFAs and BOD contents of the manure, but also that supernatant BOD appeared to be a better indicator of odor offensiveness. Other more recent studies have also confirmed a strong linear relationship between VFAs and BOD in the supernatant of swine manure (Ndegwa et al., 2002; Zhu et al., 2001; Zhu et al., 2008). From the above studies, it is quite apparent that the VFAs and BOD contents in manure supernatant can be used not only as valid indicators of odor and their respective concentrations as valid quantifiers of odor intensities, but also the two parameters can be used interchangeably. Therefore, VFAs contents and BOD contents were used as indicators to quantify odor intensities during aeration of the swine manure.

2.4 Review on the effect on oxygen transfer performance

Oxygen transfer, the process by which oxygen is transferred from the gaseous to the liquid phase, is a vital part of a number of wastewater treatment processes. Because of the low solubility of oxygen and the consequent low rate of oxygen transfer, sufficient oxygen to meet the requirements of aerobic waste treatment does not enter water through normal surface air-water interface. To transfer large quantities of oxygen that are needed, additional interfaces must be formed. Oxygen transfer performances can be affected by air flow rate, diffuser submergence, velocity in horizontal flow systems, bubble diameter, temperature, viscosity, basin geometry (affects contact time between gas and liquid), wastewater compositions (salts, surfactants, biomass) and diffuser layout and placement (Vogelaar et al., 2000; Capela et al., 2002; Gillot et al., 2005). In this part air flow rate, horizontal flow, and contaminants were taken into account.

2.4.1 Air flow rate

In clean water, an increase in the air flow rate results in a decrease in oxygen transfer efficiency (between 45 and 110 m^3/h per m^2 of membrane) (Gillot et al., 2000). Some explanations from this paper were presented as follows. This decrease is caused by an increase in the size of the air bubbles when the air flow rate increases, due to (1) a stretching of the membrane under the effect of the gas pressure resulting in an increase in the pore size; (2) an increase in the coalescence of the air bubbles.

Furthermore, the increase of the air flow rate also affects the vertical movements of the water caused by the rise of the air bubbles (spiral flows). These vertical flows, which occur between diffusers or grids of diffusers, accelerate the upward velocity of the air bubbles and are greater when the air flow rate is raised. Any increase in the air flow rate therefore results in a reduction of the residence time of the air bubbles in the water, and consequently of the oxygen transfer efficiency.

Under process conditions, an increase in the air flow rate also results in a decrease in the oxygen transfer efficiency. However, it does not always match that observed in clean water, leading to a variation in the alpha factor (ratio of the oxygen transfer coefficients in process and clean water at equivalent conditions of temperature, mixing, geometry) (Hwang and stenstrom, 1985).

2.4.2 Horizontal flow

Applying a 0.4 m/s horizontal velocity to the water induces an increase in the oxygen transfer, comprised between 40 and 50% (Déronzier et al., 1996). This enhancement is induced by an increase of the air-water interfacial area, attributed to: (1) a reduction of the nascent bubble size, due to a sharing effect of the horizontal flow on the diffusers (Da Silava- Déronzier, 1994); (2) an increase of the bubble residence time in water, due to a neutralization of the liquid vertical convection movements (spiral flows) by the horizontal flow. The first hypothesis was issued form a theoretical analysis in which coalescence of bubbles was neglected. The second one has been confirmed both experimentally (Da Silava- Déronzier, 1994) and by numerical simulations (Czarnota and Hahn, 1995; Roustan and Line, 1996).

Under process conditions, the horizontal velocity also induces an increase in oxygen transfer. The improvement observed on one site is, however, lower than in clean water (Gillot et al., 2000), leading to a decrease of the alpha factor value. This difference may be attributed to the presence of surface-active agents. These substances are derived from detergents and also produced by microorganisms (Burde and Steinmetz, 1993). Their molecule is composed of a hydrophobic section and a hydrophilic one. In solution, they tend to gather at the interface (at bubble surface in case of air injection) and form a superficial layer of orientated molecules that modify interface properties (Clift et al., 1978). The influence of surfactants on the oxygen transfer coefficient is a result of two competitive effects, on the one hand, on the liquid-phase oxygen transfer coefficient, and on the other hand, on the specific interfacial area. At the same time, they increase the resistance to transfer and make the interface rigid, both decreasing the liquid-phase oxygen transfer coefficient. The resulting effect is generally a reduction of the overall oxygen transfer coefficient in the presence of surface active agents (Hwang and Stenstrom, 1984; Gillot et al., 2000).

2.4.3 Contaminants

In comparison to clean water, the impact on oxygen transfer efficiency is modified by two groups of factors in the process water (Mueller et al., 2002; Gillot et al., 2005a): the dissolved contaminants (surfactants, inorganics) (Capela et al., 2002; Rosso and Stenstrom, 2006) and the diffuser aging process (fouling and clogging) (Wagner and Von Hoessle, 2003). The process variables, i.e., wastewater characteristics, process loading, and flow regime, influencing the concentration and distribution of contaminants have thus a potential effect on the mass transfer coefficient.

Lower flow regime gas-liquid interfaces (such as the ones produced by fine-pore diffusers) generally have a lower α than higher flow regime interfaces (such as the ones produced by coarse-bubble diffusers or surface aerators) for similar conditions (Stenstrom and Gilbert, 1981). Differences in α amongst aeration systems were noted in the 1930s (Kessener and Ribbius, 1935), but were generally forgotten until the energy crisis of the 1970s increased the awareness for energy-efficient technologies. Prior to the 1980s, many plants were designed with an α of 0.8, which was considered as a universal α for all types of aeration systems. It has been shown that different aeration methods have different α, and for fine-pore diffusers, the initial α decreases over time in operation due to fouling or scaling (Rosso and Stenstrom, 2006).

Mass transfer depression caused by contaminants has long been observed (Mancy and Okun, 1960). Eckenfelder and Barnhart (1961) reported the effects of organic substances on mass transfer, showing that contamination as low as 15 mg/L of sodium layryl sulphate (SDS) can reduce mass transfer coefficients to 0.5 times the value in clean water. The effects of wastewater contamination on mass transfer can be related to the decrease in dynamic surface tension (Masutani and Stenstrom, 1991). The interfacial accumulation of surfactants causes an increase in interfacial rigidity (hence in the drag coefficient), the reduction of internal gas circulation and interfacial renewal rates. These gas-liquid interfacial phenomena have been recently discussed by Rosso (2005).

Each type of aeration system is impacted differently by contaminants in the wastewater. Typical alpha factors achieved with various aeration devices are shown below (Table 2.1). High shear aeration devices such as jet aerators yield a higher alpha factor due to surface renewal at the gas/liquid interface. Low shear aeration devices such as fine bubble diffusers (membrane or ceramic) yield a lower alpha factor because of insufficient renewal (oxygen saturation) at the gas/liquid interface.

Table 2. 1 Alpha factor for various devices

Devices	Alpha factor
Jet aerators	0.9
Coarse bubble diffusers	0.8
Fine bubble diffusers	0.4
Surface aerators	0.85
Venturi injector	0.9

Data from www.mixing.com and www.mazzei.net

2.4.4 Temperature

Temperature stratification which develops in many lakes and reservoirs in temperate regions effectively controls vertical diffusion and thereby water quality. In the surface aeration system, the aeration depth may influence the temperature stratification.

The vertical thermal stratification observed in most water bodies is a natural occurrence, which is often characterized by a well-mixed surface layer, separated from the deeper hypolimnion layer by a strong temperature gradient, known as the thermocline. The thermocline acts as a barrier preventing active exchange of temperature, dissolved oxygen, and dissolved nutrients between the surface and bottom layer. Because of the biological and biochemical oxygen consumption, a stratified water body over a long period cannot prevent the hypolimnion from becoming anoxic leading to the increased release of undesirable substances such as nutrients, and iron and manganese from the sediment of the reservoir or lake (Kilham and Kilham 1990; Kassim et al., 1997; McGinnis et al., 2002). In order to suppress nutrient release from sediments and to ameliorate the reservoir or lake water quality, oxygen is commonly introduced into the hypolimnion artificially which is combined with artificial mixing (Yang et al., 1993; Burris et al., 2002; McGinnis and Little, 2002; DeMoyer et al., 2003).

Artificial destratification of the water column is a common means of addressing these water quality problems with the most popular method of destratification being the air bubble diffuser (McDougall 1978; Schladow 1992, 1993; Asaeda and Imberger, 1993; Yang et al. 1993; Schladow and Fisher 1995; Hornewer et al. 1997; Lindenschmidt and Hamblin 1997; Burns 1998; McGinnis and Little 1998; Simmons 1998; Johnson et al. 2000; USGS 2000; Gu and Stefan, 1995; Lawson and Anderson, 2007; Becker, Herschel and Wilhelm, 2006), which is referred to here as a bubbler. Burris et al. (2002) and McGinnis et al. (2004) demonstrated via experiment that the hypolimnetic oxygenation can be achieved by injecting oxygen into the hypolimnion using oxygen bubble diffuser without disrupting thermal stratification. The oxygen bubble diffuser system does not dismantle the thermal stratification; thus, the thermocline strongly inhibits the vertical mixing, and thereby cuts off the flow of DO from oxygen rich epilimnion to hypolimnion layer. The surface water in contact with air becomes in equilibrium with the atmospheric oxygen concentration and ceases further oxygen dissolution into the water from the atmosphere. Therefore, the total amount of gaseous oxygen dissolution from atmosphere into water is significantly reduced. On the other hand, the overall goal of the air bubbler is to sufficiently reduce the stratification so that the water body may completely mix under natural phenomena and remain well oxygenated throughout (Johnson et al., 2000). Unfortunately, there is little information about the effect of artificial surface aeration or circulation on vertical dynamic temperature changes in a surface aerated setting.

Chapter 3 Venturi aerator module development

3.1 Introduction

There have been reports indicative of promising results in using aeration for odor control (Williams et al., 1984; Williams et al., 1989; Pain et al., 1990; Sneath et al., 1992; Zhang et al., 2004; Zhang and Zhu, 2005). Under aerobic conditions, biodegradable organic materials such as swine manure can be oxidized into stable inorganic end products by aerobic bacteria that require oxygen (Westerman and Zhang, 1997). In this process, the nitrogenous compounds are oxidized to nitrite and then to nitrate. In that regard, use of aerobic processes (as opposed to anaerobic) enables the management of excess nitrogen as di-nitrogen (N_2) gas through localized denitrification (Burton et al., 1993). In addition, odorous compounds such as sulfide and mercaptan are also decomposed to form odorless sulfate (Westerman and Zhang, 1997). Work by Williams (1984) has quantified the relationship between the offensiveness of odor and the volatile fatty acids (VFAs) concentration in treated pig slurry, which is a major group of odorous compounds that can be controlled by aerobic treatment. Williams et al. (1984, 1989) also showed how the return of an offensive odor in stored aerobically-treated liquid manure, indicated by the increase in VFAs content, was determined by the aerobic treatment regime that the manure had undergone prior to storage. All this information clearly demonstrates that aeration (aerobic treatment) has been proved to be an effective technique in manure odor control.

Despite such encouraging and promising information, utilization of aeration to control odor at farm level does not appear to progress. The major hindrance of using this technique rests with the fact that most of the commercial aeration systems are expensive relative to production costs, i.e., for a better effluent with low organic strength, the treatment costs tend to go higher. Conversely, the low-cost options tend to equate with less, and sometimes inadequate, treatment. This constitutes the reason why the tremendous work done by the past researchers has all ended up with just one step short in transferring this technique into field-scale implementation. Poor aeration is mainly due to the inefficiency of oxygen transfer by the aerators when used in large-scale operations. Therefore, to overcome this problem and advance the knowledge for field application, there is an acute need to develop better aerators that are able to increase aeration efficiency without increasing energy consumption. Only in this way can aeration become a truly cost effective method for animal producers to control odor. The objective of this research was to evaluate the performance of developed venturi aerator modules according to oxygen transfer efficiency. This research presented information on design and evaluation of six modules composed of venture aerators and determining the best aeration efficiency and oxygenation capacity. And the oxygen transfer coefficient (k_La), the standard oxygen transfer rate (SOTR), and the standard oxygenation efficiency (SOE) for each configuration were ascertained and compared based on clean water tests.

3.2 Materials and methods

3.2.1 Aerator module design

Studies on aeration systems include theoretical and experimental methods. Based on the mass transfer method and tests conducted in the laboratory, one, two, and three venturi aerators connected either in series or in parallel were developed and evaluated. Six different venturi aerator modules (Figure 3.1) were analyzed in this research aimed at efficiently transferring oxygen into water. Both configurations were considered capable of adding more oxygen into liquid than when only one venturi air injector was used. The actual number of aerators used in each configuration was determined by water tests according to the maximal oxygen transfer coefficient achieved under the respective conditions. The air was entrained for series design (Figure 3.1 a, b, c) in a stepwise manner by each in-line venturi air injector until a point was reached when addition of more aerators to the module would cause a decline in oxygen transfer efficiency; a result of the liquid pressure drop across the injector module resulting in reduced overall liquid flow rate, thus reduced amount of oxygen transferred by each injector into the liquid. In addition, the increased oxygen content in water by early injectors would reduce the transfer efficiency by the later ones due to the reduced deficit between the oxygen concentration in the air and liquid phases. By using this approach, the optimum oxygen transfer efficiency could be determined.

For the parallel design (Figure 3.1 d, e, f), the pressure drop of each aerator is identical and the sum of the liquid volume of each aerator is the total liquid volume. The transfer of oxygen was perceived to be evenly shared by all air injectors in the aerator module, with each injector only responsible for aerating a portion of the total liquid volume. The reduced liquid volume flowing through each individual air injector allowed the use of smaller injectors in order to maintain the needed liquid flow rate for good air entrapment. Generally speaking, small nozzle diameter injectors have advantages over large ones in that they can produce finer air bubbles (atomization) and better mixing,

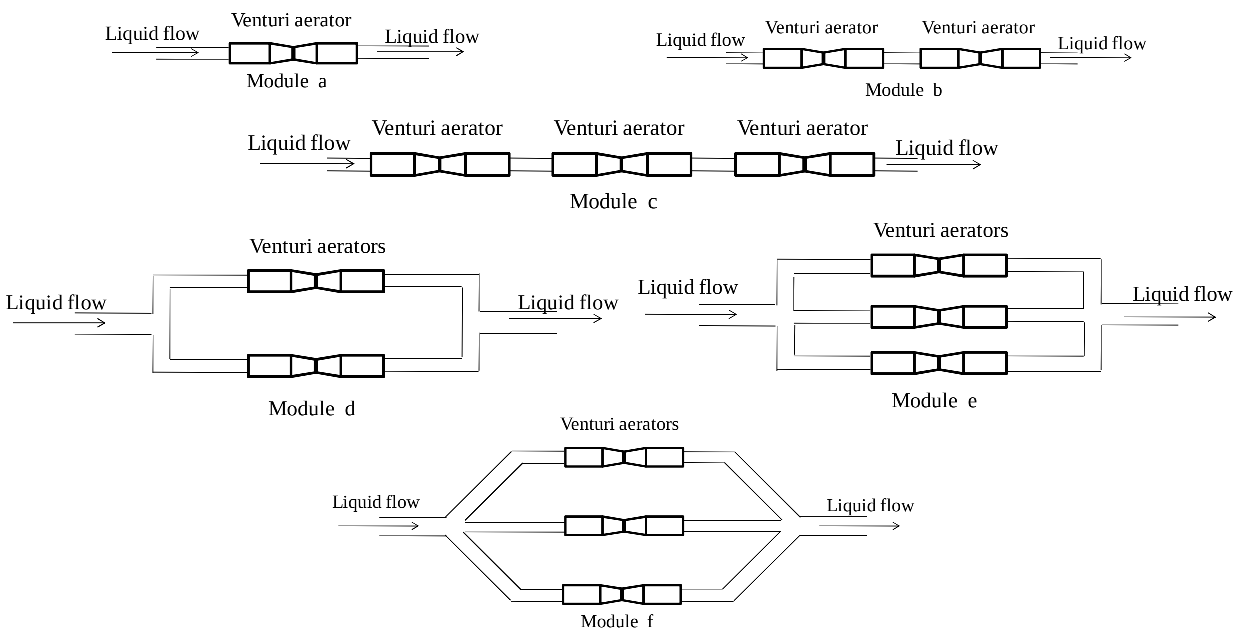


Figure 3. 1 Schematic of the six configurations of the air injector module design

thereby enhancing the oxygen transfer efficiency. The factor limiting the use of small injectors in large wastewater treatment facilities was the throughput capacity. Fortunately, the parallel design with multiple air injectors could solve that problem. Also, limited research information (our unpublished data) indicated that the parallel design could be operated by a smaller pump than needed for the series design, potentially reducing power consumption. This, however, needs to be further verified. Similar to the tests for the series design, the number of aerators needed to form the aerator module with the optimal oxygen transfer efficiency was determined based on the maximal oxygen transfer coefficient achieved. The injectors in the parallel design was placed 20 cm apart from each other. It is easier to determine and maintain the best work conditions for venturi aerators. According to the pump size and the best work conditions of each aerator, we can determine the number of aerators used in the aeration system.

All venturi aerators used in this project were purchased from the Mazzei Injector Corporation (model#: 1583; thread diameter: 3.8 cm; 500 Rooster Drive, Bakersfield, CA 93307).

3.2.2 Design of apparatus for the aeration efficiency tests

The overall apparatus for evaluating the new aerator modules is shown in Figure 3.2, which comprises a 4.55 m^3 (1200-gallon) water tank, an aerator module, a 1.5 kW (2-hp) centrifugal water pump (liquid flow rate: 3.4 L/s), a data acquisition system, dissolved oxygen (DO) probes, and piping materials. When operating, the centrifugal water pump circulated the liquid through the venturi air injection module to entrain air into the liquid. To maximize air retention time in the liquid, two perforated PVC pipes (2.5 cm in diameter), joined perpendicular to each other, were placed horizontally at the bottom of the tank so the oxygenated liquid was always dispersed into the tank at the bottom in order to reach a uniform oxygen distribution in the water. Openings (0.6 cm (1/4") in diameter) were made on the pipes at 5-cm intervals; some facing up while the others horizontally to evenly distribute the oxygenated liquid in the entire tank. For each test, roughly 2.28 m^3 (600 gallons) of water was used for determining oxygen transfer coefficients.

The detailed clean water test procedure is as follows:

1. Cleaning of the tank and the aeration devices was done thoroughly and the instruments to be used for measurements of various parameters were calibrated before the start of experimentation. The calibration of DO meter was done

according to the procedure described by the manufacturer and by using the standard solutions of potassium chloride (KCl), sodium sulphite and distilled water provided by the manufacturer. This procedure was repeated for all sets of experiments.

2. The tank was filled with tap water available in the laboratory up to 2.28 m^3 (600-gallon). The water volume for every experiment was maintained constant.

3. The DO is reduced to zero by oxidizing the water with sodium sulfite catalyzed by cobalt. The sodium sulfite is added in amounts usually equal to 125 or 175% of the stoichiometric requirement and cobalt chloride is added to produce a concentration of approximately 0.05 mg/L as cabaltous ion. It took about 10-15 minutes to deoxygenate the water, which could maintain DO between 0.0-0.1 mg/L for about 5 minutes afterwards.

4. Increase in DO concentration was measured by DO meter at the water surface and was recorded at 1 minute intervals after switching on the aerators. Such measurement continued using a DO probe until the DO concentration increased from 0% to at least 90% saturation.

5. Simultaneously the effluent liquid flow rate and temperature were also recorded by using a flow rate meter and thermometer, respectively.

The same procedure was repeated for all the aerator modules under test for their performance evaluation with respect to the overall oxygen transfer coefficient by changing the various parameters.

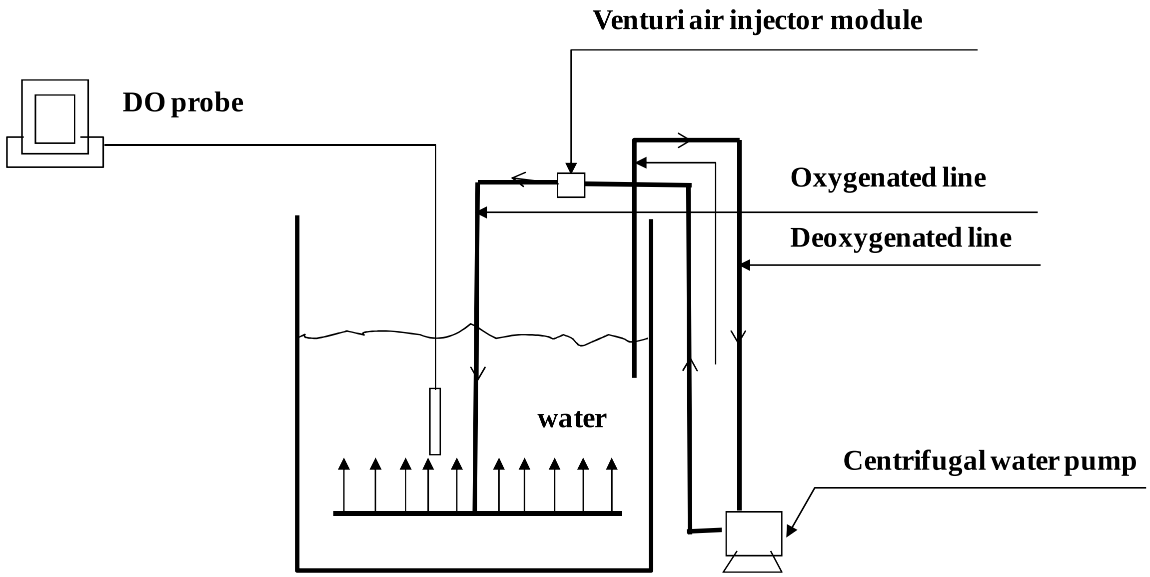


Figure 3. 2 Schematic of the apparatus for determining aeration efficiency of the aerators

3.2.3 Gas transfer

The performance of particular aerators should be determined under the conditions that the aerators are to be used. However, due to the complications introduced by the microorganisms present in the wastewater, aerators are normally evaluated in clean water with other operating conditions similar to real conditions, such as temperature, volume of water to be aerated, and dimensions of aeration vessel. Using a standard method provided by the American Society of Civil Engineers (ASCE), different aerators modules were tested in clean water to determine SOTR and SOE. The apparatus presented in Figure 3.2 was used for the clean water experiments to determine the optimal aeration efficiency with respect to the number of venturi air injectors used in the system. Prior to each test, the water was deoxygenated by adding sodium sulfite

(150 mg/L) and cobalt chloride (1 mg/L). Once the dissolved oxygen (DO) level in water reached zero, aeration would start and the DO level in the water was recorded at 1-minute intervals until it approached saturation. The test was repeated three times and the mean values for all measurements were presented. In addition to DO measurements, water temperatures were also recorded for each individual run using a thermometer located in the center of the water tank.

Non-steady state methods were employed in both tests for determining the oxygen transfer coefficient, k_La, (Eckenfelder and Ford, 1968). The variation of the oxygen concentration in the water, as a function of time, is given as follows:

$$\frac{dC}{dt} = k_L a(C_\infty - C) \tag{1}$$

Where C is the oxygen concentration in water (mg/L); C_∞ determination point value of the steady DO concentration at time approaches infinity (mg/L); t the time (h); k_La the overall mass transfer coefficient (h^{-1}).

Integrating the equation between the limits $C=C_0$ at time t=0 and $C=C_t$ at t=t, k_La can be determined from the slope of a semi-logarithmic plot of the concentration deficit (C_∞ – C) versus time of aeration. The equation can be given as:

$$(k_L a)_T = \frac{\ln(C_\infty - C_0) - \ln(C_\infty - C_t)}{t} \tag{2}$$

Where ln represents natural logarithm of the given variables and the concentration C_∞, C_0, and C_t are expressed in mg/L.

The k_La is a function of temperature. In order to compare the coefficients for different temperatures, the following equation can be used to adjust the coefficients under different temperatures to standard condition (T=20 °C).

$$(k_L a)_{20} = \frac{(k_L a)_T}{1.024^{(T-20)}} \tag{3}$$

Where $(k_La)_{20}$ is the overall mass transfer coefficient at T=20°C (h^{-1}); (k_La)T the overall mass transfer coefficient at the water's temperature for the test (h^{-1}); T the water temperature (°C).

Nonlinear regression is employed to estimate k_La and C_∞. These estimates are adjusted to standard conditions (20°C water temperature, zero DO concentration and one atmosphere). The SOTR is obtained as the average of the products of the adjusted determination point k_La values, corresponding to the adjusted determination point C_∞ value, and the tank volume. And the SOTR value was divided by the power consumption applied to obtain the SOE.

$$SOTR = k_L a(C_{\infty}{}^{*}{}_{20})V \tag{4}$$

Where k_La is the determination point value of k_La corrected to 20 °C; $C_{\infty 20}$* the determination point value of steady-state DO concentration corrected to 20 °C and a standard barometric pressure of 1 atmosphere; V the liquid volume of test water in the test tank.

3.3 Results and discussions

3.3.1 Variation of DO with time for different aerator modules

Data from three runs of six aerator modules are averaged and presented under standard conditions in Figure 3.3, which depicts the changes in dissolved oxygen (DO) concentration during aeration. The aerator module in parallel gave a better oxygen transfer capability than that in series in view of the fact that at any sampling point after aeration started, it transferred much more oxygen into the test liquid, as shown in Figure 3.3.

For module a, b and c in series, the DO provided by module a was always the highest and the oxygen transfer efficiency appeared to deteriorate as the number of air injectors increased in the system. There was no big difference of oxygen transfer efficiency between module b and c, so increasing the number of aerators was not justifiable. This observation suggested that further testing of the module with four air injectors was not necessary, although the physical construction of the module was completed. During the experiment, the module with one air injector produced the highest air suction possibly because of the minimal pressure drop existing between the inlet and outlet of the injector. When two injectors were tested, an obvious reduction in air suction for both injectors was recognized. The increase in resistance to liquid flow in the two-injector module could reduce the flow rate, evidenced by the drop in pressure difference across the first injector in the two aerator module from 137.8 to 133 kPa. This decrease could lead to reduced speed of liquid passing through the air suction port and thus less air entrapped, a critical part in oxygen transfer efficiency.

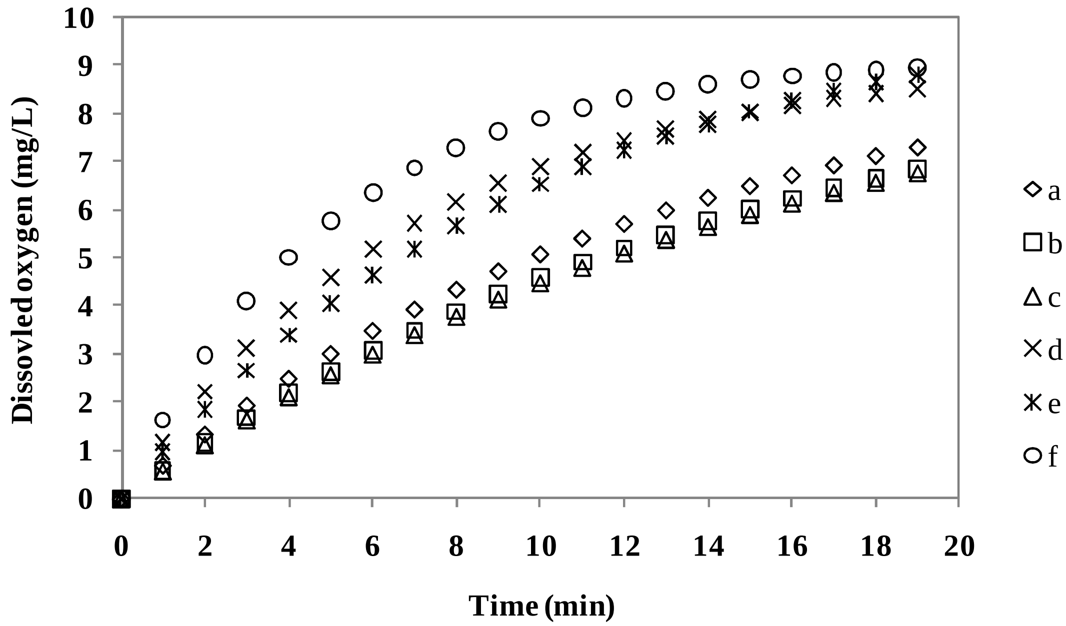


Figure 3. 3 Changes in the mean DO concentration during the aeration period for module a, b, c, d, e and f under standard conditions

When the number of injectors increased to three, it was observed, by using hands to feel the air suction at each port, that only the last one along the liquid flow direction was actually trapping air into the system, and at a much lower capacity (drop in pressure difference across the first two injectors to 82.1 kPa). This is why the dissolved oxygen curve hits the lowest as illustrated in Figure 3.3. As the pressure dropped across the aerators and resistance increased in series design, air suction was reduced through the aerators. So module c with three aerators didn't work well and entrapped less air than other modules in series.

Setups with two and three aerators in parallel were able to bring more oxygen into water than modules in series. From Figure 3.3, aerator module d with two air injectors gave a better oxygen transfer capability than module e in the first 12 minutes after aeration started, and it transferred more oxygen into the test liquid than module e, with the differences in DO levels in water ranging from 0.13 to 0.50 mg/L over the experimental period. For module f, with the reduced friction and a more even flow rate in each line, it was the most efficient in terms of oxygen transfer among all the modules investigated, considering that the same test conditions were used. Compared with module e, module f can gain more DO levels in water ranging from 0.21 to 1.68 mg/L over the experimental period. Maximizing the oxygen transfer from the gaseous phase to liquid phase using minimal energy is the fundamental goal of efficient aeration.

3.3.2 Effect of the designs on oxygen transfer efficiency

A semi-log graph based on Equation 2 that reflects the most important parameter in evaluating any aeration system in terms of efficiency was presented in Figure 3.4, in which the slope of each linear regression line is the k_La for that particular aerator module. Since a non-linear regression (exponential) was used for curve fitting, the values of k_La could be read directly from Figure 3.4 for each aerator module. In this case, the k_La values under standard conditions for module a, b, c, d, e and f 0.076 min^{-1}(4.54 h^{-1}), 0.063 min^{-1}(3.79 h^{-1}), 0.060 min^{-1}(3.58 h^{-1}), 0.139 min^{-1}(8.37 h^{-1}), 0.099 min^{-1}(5.93 h^{-1}), and 0.198 min^{-1}(11.87 h^{-1}), respectively. Module f showed the best oxygen transfer coefficient. A comparison of k_La was presented in Table 3.1. The one aerator module in the series design showed the best oxygen transfer coefficient, while there was not much difference between using two or three venturis in series.

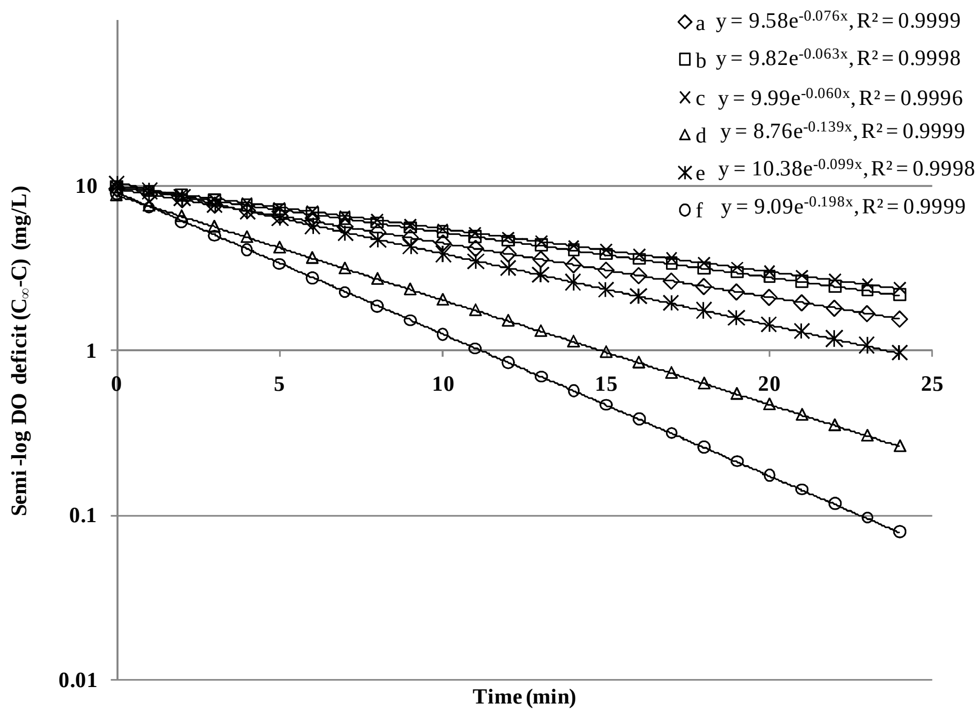


Figure 3. 4 Determination of oxygen transfer coefficients for module a, b, c, d, e and f under standard conditions

For practical purposes in aeration of slurries and wastewaters, the energy requirement of the aeration equipment used to transfer oxygen in the liquid usually represents 60% to 80% of the wastewater treatment operating cost (Vaxilaire et al., 1995). The fundamental goal of efficient aeration therefore is to maximize the dissolution of oxygen into the liquid from the gaseous phase using minimal energy. On that account, module f is the best among both the parallel and series modules investigated. Given the great deal of engineering effort and resources spent in developing new aerators to enhance oxygenation capacity, the findings from this study provide a different perspective in improving aeration efficiency through innovative designs using inexpensive, commercially available aerators. Further work in this area should be pursued.

Based on k_La, the SOTR and SOE for all the aerator modules were determined. The SOE for module a, b, c, d, e and f were 0.07, 0.06, 0.06, 0.12, 0.10, and 0.21 kgO_2/kWh, respectively. The aerator comparison with respect to SOE showed the highest value for module f and the lowest for module b and c. And for the modules in series, use of more aerators deteriorated the aeration efficiency. On this account, parallel design worked better in using multiple venturi aerators. For SOTR, module f outperformed other designs by large margins (0.31 kgO_2/h).

Table 3. 1 Performance of all modules

	T (°C)	$k_La(T)$ (h^{-1})	$k_La(20)$ (h^{-1})	$C_\infty^*(T)$ (mg/L)	$C_\infty^*(20)$ (mg/L)	SOE (kgO_2/kWh)	SOTR (kgO_2/h)
Module a	15.2	4.05	4.54	10.58	9.58	0.07	0.10
Module b	16.1	3.45	3.79	10.64	9.82	0.06	0.09
Module c	16.7	3.31	3.58	10.69	9.99	0.06	0.09
Module d	13	7.09	8.37	10.63	9.17	0.12	0.18
Module e	17.9	5.64	5.93	10.83	10.38	0.10	0.15
Module f	23.6	12.92	11.87	8.48	9.17	0.21	0.31

By comparing the results reported herein for the different modules, it can be understood that better aeration efficiencies can be achieved by simply changing the way the venturi aerators are connected. And module f achieved the highest performance among all the modules tested, with the SOTR and SOE increased by 3- and 3.5-fold compared to other five designs. However, the performance of the tested six venturi aerator modules showed lower SOTR and SOE than the published literature values for typical mechanical aerators including venturi aeration systems (0.13 to 3.85 kgO_2/kWh)

(Cumby, 1987). Zhang et al. (2007) evaluated six types of aerators (including Vortitech, Black Circulator, Pond Mill, VBT TM, AIRE-O2 Aerator, and AIRE O2 Trition) in clean water. The SOTR was determined to be 0.20, 0.15, 0.21, 2.66 and 2.93 kgO_2/h, respectively, while the SOE was 0.49, 0.35, 0.36, 0.10, 0.68 and 0.57 kgO_2/kWh. Compared to these results, the performance of venturi aerator modules tested in this study was on average. Therefore, aerators currently used for manure wastewater lagoons need significant improvement in their performance and more work is needed to further the effort in this area to develop cost effective and energy efficient aerators for the applications on livestock farms. Nonetheless, the results reported herein have fulfilled the objective of this project by providing evidence that better aeration efficiencies can be achieved by innovative designs of aerator modules using low cost venturi air injectors.

Another point of interest rests in the inferior performance of the aerator module with three parallel-connected venturi injectors as compared to the one with two. According to the information from the injector vendor (Mazzei Injector Corp., Bakersfield, Calif.), the particular model used in this study requires a liquid flow rate of 1.2 L/s to obtain the best air suction capability. In our design, the total maximal liquid flow rate that can be provided by the pump is 3.4 L/s, indicating an insufficient liquid flow rate for each air injector, assuming even distribution of liquid among three injectors. This may explain the overall loss of aeration efficiency in the system. However, since the liquid flow rates were not measured for each injector, the actual reduction in air suction could not be individually determined for the aerator module with three parallel-connected injectors.

3.4 Conclusions

Six configurations of venturi aerator modules were evaluated by determining their k_La, SOTR and SOE using the standard method of ASCE for testing aerators in clean water. For the series design (module a, b and c), oxygen transfer coefficients of 4.54, 3.79, and 3.58 h^{-1} were observed, respectively. While for the parallel design (module d, e and f), the corresponding values were 8.37, 5.93 and 11.87 h^{-1}. Module f achieved the highest oxygen transfer coefficient among all the aerator modules tested.

The SOTRs for the six aerator modules were determined to be 0.10, 0.09, 0.09, 0.18, 0.15, 0.31 kgO_2/h and the SOEs were 0.07, 0.06, 0.06, 0.12, 0.10, and 0.21 kgO_2/kWh, respectively. All the aerators tested showed lower SOTR and SOE than the published literature values for typical mechanical aerators. SOTR of module f increased by 3-fold and SOE by 3.5-fold compared to other five designs.

Data from this study have shed light on the likelihood of improving the aeration efficiency of a venturi type system by innovative aerator module designs, and paved the way for continued research to significantly enhance the performance of current venturi aerators in order to match the cost-effective requirement. More aerator module development and research is needed to improve the oxygen transfer efficiency of such aeration systems so effective control of odor from liquid swine manure lagoons can be achieved. The improved aeration system can also be applied to manure lagoons for other livestock species as well.

Chapter 4 Use surface aeration to control odor at lab and field scales

4.1 Introduction

Not many farms are using surface aeration to control odors from the manure storages or lagoons. The major hindrance of using this technique rests with the lack of field data because all the results or conclusions presented by previous researchers are solely based on either laboratory or quasi-pilot scale tests with little useful information that can be utilized to promote field evaluation and application. As such, the knowledge gap between lab studies and field application has not been bridged, which constitutes the reason why the work done by the past researchers has all ended up with just one step short in transferring this technique into field-scale implementation. To overcome this shortfall and advance the knowledge for field application, there is an acute need to carry out field-scale experiments as a means of verifying the lab-scale experiments, thus paving the way for on-farm applications.

In this chapter, a lab-and-field scale study was conducted to reveal the effect of surface aeration treatment on reducing odor generation potential from manure storages based on the changes of characteristics of two chosen indicators, i.e., BOD and VFAs. Besides, DO, pH, TS, TVS and nitrogen components were also evaluated and the correlation among these parameters was analyzed. The information from this work is essential to prove that surface aeration can be an effective tool in reducing odor generation potential from lab-scale especially field-scale open manure storage structures.

4.2 Experiment on surface aeration to reduce odor generation potential at lab scale

4.2.1 Materials and methods

Fresh swine manure was drained from a pull-plug finishing barn to a collection sump and then pumped through a solids-liquid separator (sieve opening: 2.5 mm). The separated liquid was used for this study. The initial manure level, with 0.74% TS content, in both the aerated and the control tanks was 93 cm. For manure loadings, fresh manure was diluted with tap water to approximately 1.0% to mimic the solids content in the liquid after solids liquid separation treatment and was added, from the top, to both tanks once a week. This operation simulated a pull-plug manure handling system where fresh manure in the shallow pit was flushed out to the outside storage every week.

Liquid manure samples were collected from the center of the aerated layer in the tank twice in the sampling week (before and after the loading day), and the sampling was carried out every two weeks. The collected manure samples were analyzed according to the Standard methods (APHA et al., 1998). The concentration of VFAs was measured according to Method 8196, DR/2800 spectrophotometer Manual (Hach Co., 2005). Whenever analysis was not done immediately after sampling, the samples were stored at -20°C and then thawed and allowed to reach room temperature prior to analysis.

4.2.2 Experimental setup

The experimental design was presented in Figure 4.1. Two circular concrete tanks (2.4 m in diameter and 2.4 m in depth) were used as pilot-scale manure storage ponds: one for treatment and the other for control. For the aeration tank, two crisscross PVC pipe structures were placed about 61 cm apart along the depth with the liquid suction crisscross located at the top. A rigid PVC pipe vertically connected the two crisscrosses together. Openings (0.64 cm in diameter) were made on the pipes at an interval of 10 cm, either facing down (for the liquid suction pipes) or facing horizontally (for the liquid injection pipes). The aeration system for the aeration tank consisted of a venture air injector, a 0.75-KW centrifugal pump (Environmental systems, Technology and Research, Inc., Brussels, Wis.), and some flexible hoses that connected the PVC pipe frame to the air injector.

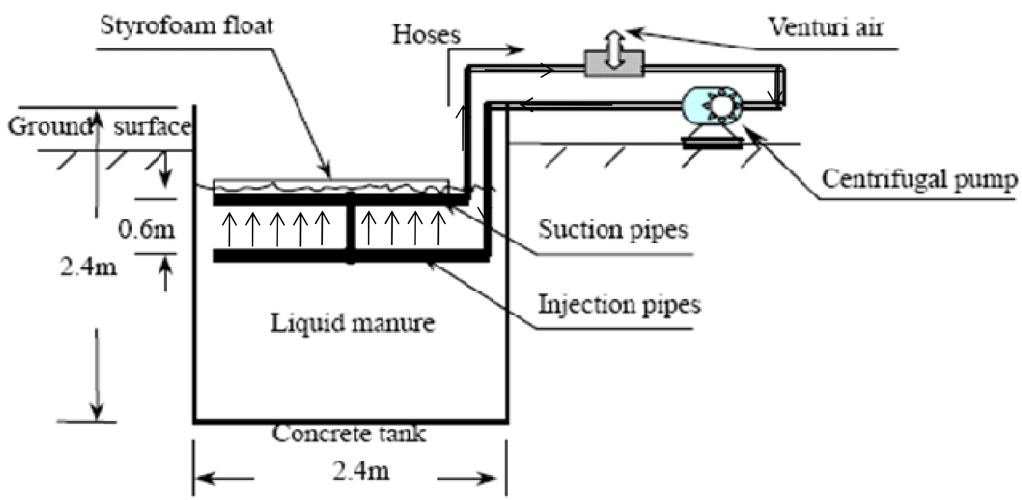


Figure 4. 1 Schematic of the apparatus for determining odor generation potential reduction efficiency of the surface aeration

According to the specification from the venturi aerator manufacturer, the centrifugal water pump was fixed by maintaining the water pressure at 1 atm to entrain air into liquid. The temperature was not controlled during the test. Styrofoam floats were mounted on the top PVC pipes so the whole PVC pipe structure could move up and down with the liquid level in the tank. This design can maximize air retention time in the liquid in that the oxygenated liquid was always dispersed into the tank from the bottom up, with the deoxygenated liquid in the top pipes being sucked into the outside system for air entrapment.

4.2.3 Results and discussions

4.2.3.1 DO

Figure 4.2 presents the DO levels measured at 10 A.M. (2 h after the centrifugal pump was turned on) three days a week. According to Figure 4.2, the DO levels in the aerated liquid manure were in general higher than 1 mg/L and increased gradually in the course of aeration treatment, while the average DO levels from the control were 0.2 mg/L. The fluctuation in the DO levels reflects the manure loading pattern, i.e., the first DO reading after the loading day was always low because of the addition of fresh organic materials that might increase the aerobic activities, thus depleting oxygen in the liquid.

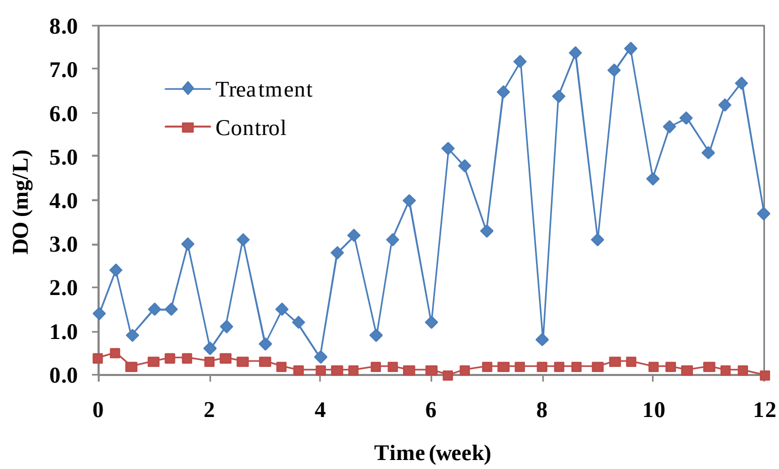


Figure 4. 2 The dynamic changes of DO in liquid manure measured at 10A.M. three times a week

With the organic compounds being decomposed, the DO level in the liquid apparently was recovered to the level prior to manure addition, featuring a dynamic oxygen uptake process commonly seen in the aerobic treatment process. Similar observations were also reported by Tripathi and Allen (1999). Although the DO levels in the aerated tank fluctuated between 0.4 and 7.5 mg/L, the data points showed that 80%, 68%, and 32% of the DO measurements were higher than 1, 2, and 5 mg/L, respectively. Our surface aeration system apparently is able to increase the oxygen concentration in the manure to a range between 10% and 60% of the saturation concentration based on the operating scheme in this study.

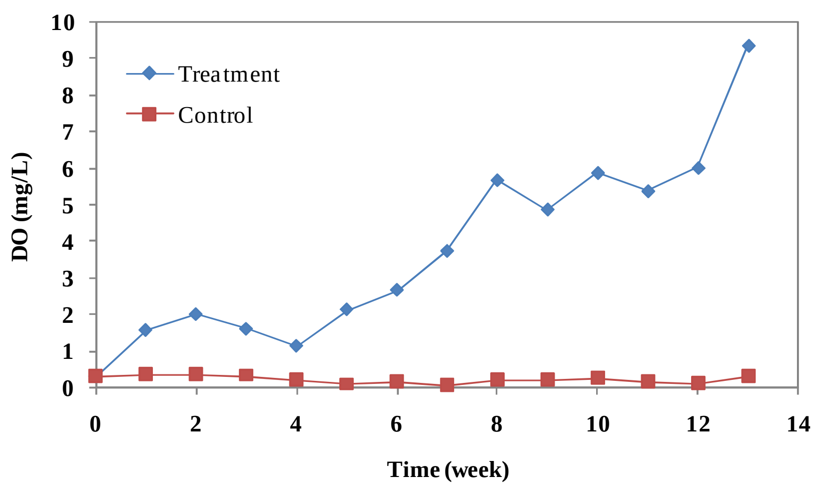


Figure 4. 3 The weekly averaged dynamic changes of DO in liquid manure measured at 10 A.M.

Figure 4.3 presents the averaged DO levels measured once a week (averaging the data from Figure 4.2). According to Figure 4.3, the DO levels in the controlled liquid manure were in general stable ranging from 0.1 to 0.2 mg/L in the course of experimental period; while the DO levels from the aeration treatment were increased gradually. The fluctuation in the DO levels reflected the difference in the oxygen supply rate and the biological consumption rate (WPCF and ASCE, 1988). The DO concentration obviously increases whenever the oxygen supply exceeds the biological consumption rate, remains constant when the two rates are equal, and decreases when the consumption rate exceeds the supply rate. From Figure 4.3, it can be seen that the oxygen supply exceeded the biological consumption rate after the aeration operation and the aerobic bacteria were established.

Figure 4.4 and 4.5 represent the biweekly dynamic changes of DO levels in liquid manure measured at 8 A.M., 2 P.M. and 6 P.M. before and after loading day. From Figure 4.4, it can be found that in the first 6 hours of aeration period, the oxygen supply rate exceeded the consumption rate and DO concentrations increased. And then in the next 2 hours, the oxygen supply rate was almost equal to the consumption rate. Therefore, the DO concentrations almost remained constant from 2 P.M. and 6 P.M. It implied that the aerobic bacteria played an important role in the system.

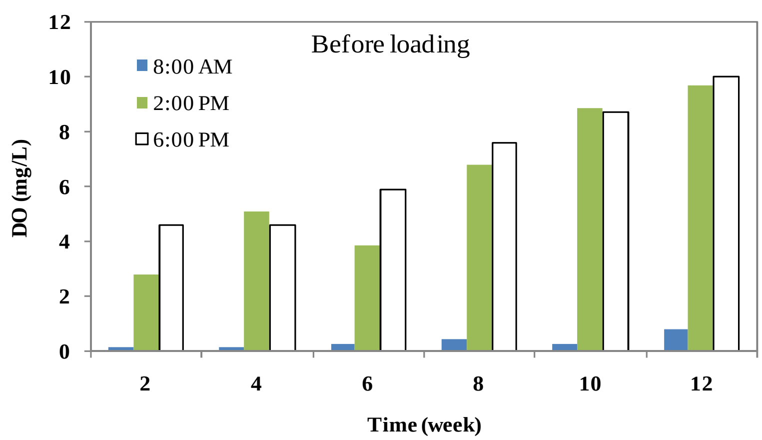


Figure 4. 4 The biweekly dynamic changes of DO in liquid manure measured at 8 A.M., 2 P.M. and 6 P.M. before loading day

From Figure 4.5, it can be found that in the entire aeration period, the oxygen supply rate exceeded the consumption rate and DO concentrations increased. Comparing the data of 2 P.M. from Figure 4.4 and 4.5, it implied that addition of fresh organic materials increased the aerobic activities, thus depleting oxygen in the liquid, inducing DO concentrations that were lower after the loading day than those before the loading day. Because of the stable aerobic system, it just took a short time that the DO concentrations were recovered and increased linearly after the addition of the fresh swine manure. It implied that efficient surface aeration can build stable aerobic bacteria playing an important role to reduce odor generation potential.

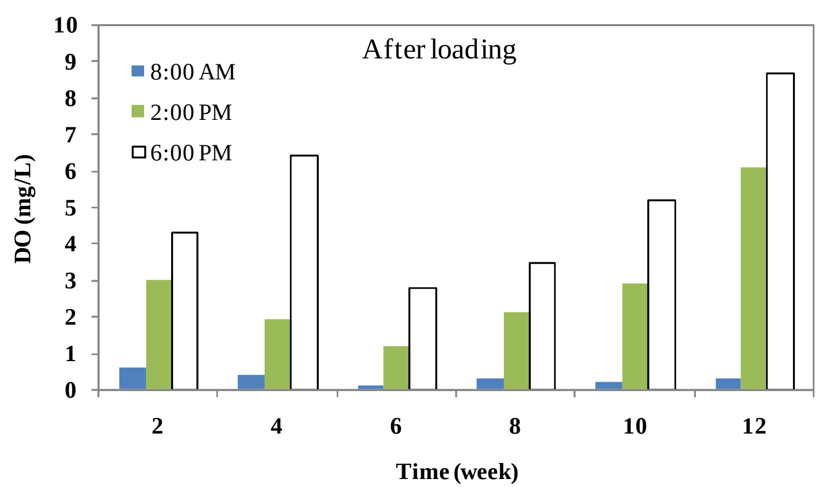


Figure 4. 5 The biweekly dynamic changes of DO in liquid manure measured at 8 A.M., 2 P.M. and 6 P.M. after loading day

4.2.3.2 pH

Figure 4.6 represents the weekly averaged pH levels in the surface aeration system under treatment and control. Surface aeration sharply increased liquid manure pH within the first month by more than one unit, from 6.9 to 8.3, and then maintained the pH around 8.2 for the rest of the test period. Researchers have found that odors can be reduced in varying degrees when manure pH is raised to a range from 8 to 11 (Vincini et al., 1994; Bundy and Greene, 1995). One major reason that the raised pH may reduce odor is that it inhibits the growth of odor-causing bacteria indigenous to swine manure (Zhu, 2000). Obviously, the effect of increasing pH caused by the surface aeration system studied should be beneficial to odor generation potential reduction.

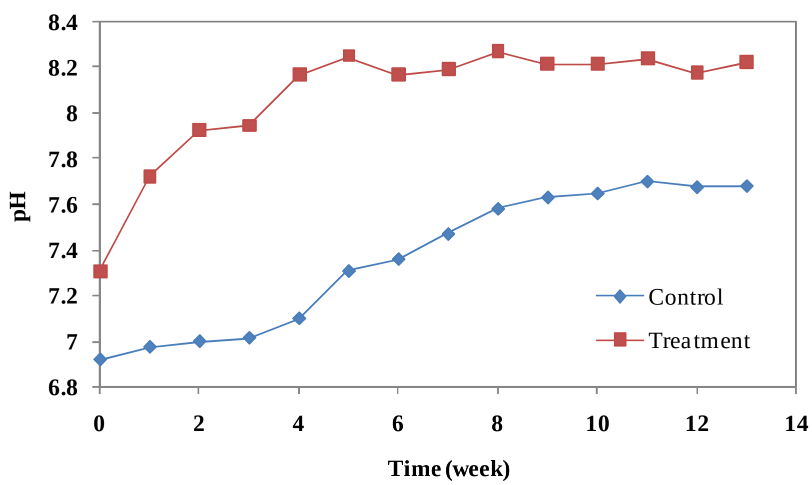


Figure 4. 6 The pH levels changes under aeration treatment and control

4.2.3.3 Change of solids composition

Figure 4.7 represents the changes of TS, TVS levels and the ratio of TVS/TS. Because the solids content of the fresh liquid manure added weekly was controlled at 1%, the daily changes of TS and TVS contents were not significant in the unaerated tank, regardless of sampling time. With the increase in storage time and the amount of manure in the tank, a trend that solids levels slowly decreased was found. This trend, in part, may be attributed to solids settling besides the breakdown of suspended organic matter by the natural metabolic actions of microorganisms.

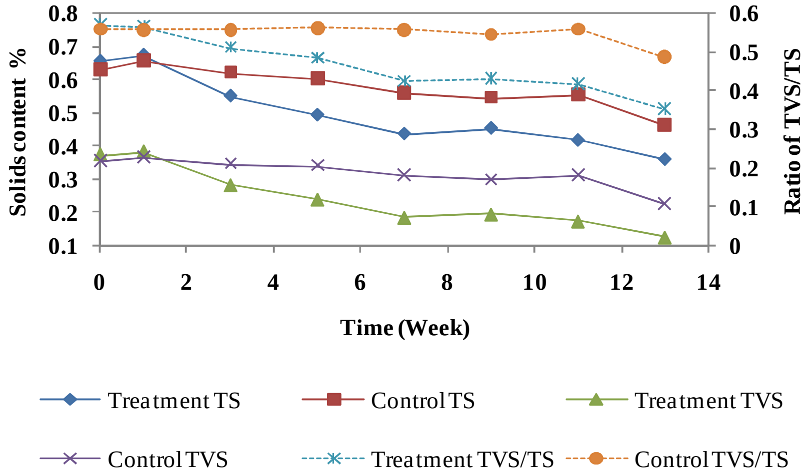


Figure 4. 7 The changes of TS, TVS and TVS/TS under aeration treatment and control

As to the manure under surface aeration treatment, there were at least four major observations. First, the daily changes of solids content in the 13 storage weeks were negligible. Second, in the first two weeks of the experiment, no difference was observed for the solids content between the two tanks. However, rapid decreases in solids content were observed after two weeks of aeration treatment, particularly for the volatile solids. Third, compared to the unaerated treatment, the dynamic solids removal efficiency increased from 9.26% to 23.20% for TS and 15.53% to 45.78% for TVS. Moreover, the TVS removal efficiency was generally higher than that of TS. The removal efficiency difference between TVS and TS increased from 6.72% to 25.58% and from 8.48% to 20.37%. The ratio of TVS to TS decreased from about 0.57 to 0.35 after 13 weeks aeration treatment, while no significant changes were found in the unaerated manure. The change in the ratio of TVS to TS appeared not to be influenced by the addition of fresh manure, which implied that the treatment was relatively stable.

Volatile solids generally refer to the various organic components with higher biodegradability than the rest of solids that is mainly associated with the coarse material in the slurry and effectively inert in the typical aerobic treatment process (Burton, 1992). Combining the fact of increasing removal efficiency of TS and TVS with the trend of decreasing percentage of TVS/TS, it is obvious that both the quantity and intensity of the readily decomposable solids related to odor generation potential are markedly reduced by the surface aeration treatment.

4.2.3.4 BOD

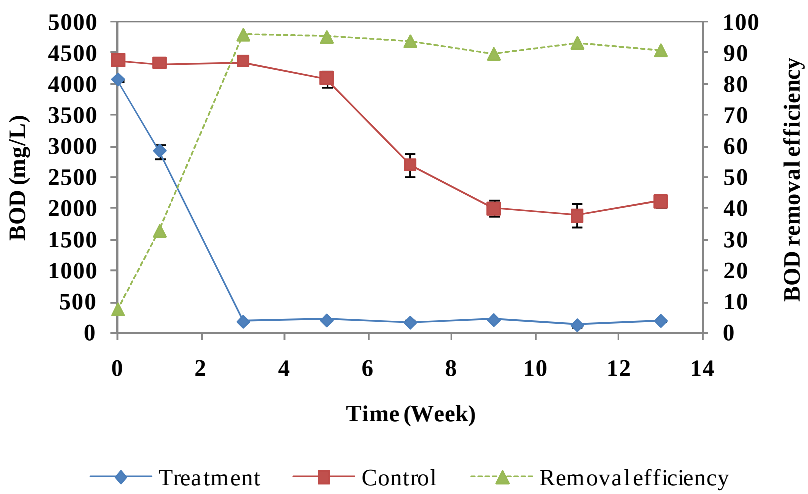


Figure 4. 8 The changes of BOD and removal efficiency

The dynamic changes in BOD content biweekly is shown in Figure 4.8. In the first three weeks (the results from the first sampling day was considered as the initial data labeled as week 0), BOD levels were drastically reduced linearly in the aerated manure. After that, the difference of BOD levels between the two treatments was around 4000, 2500, and 2000 mg/L at week 6, 8, and 10, respectively. And after the first three to four weeks, BOD levels was stable and low despite that the BOD levels in the fresh manure added weekly were generally higher than 3200 mg/L. This indicated that the system had successfully established an aerobic bacterial consortium in the top aerated layer after about three to four weeks of operation. This bacterial group can actively decompose newly added manure without fail because most of the newly added BOD from the fresh manure was apparently consumed in the aerated tank within several hours after the system was operating in a steady state.

4.2.3.5 VFAs

The volatile fatty acids are the intermediate products generated during the microbial decomposition of manure (Zhang et al., 1997). The key to preventing odor generation is that the production of acids by the indigenous bacteria and the consumption of acids by the methanogens to produce methane and carbon dioxide have to be in equilibrium (Zhu, 2000). The effectiveness of VFAs removal by surface aeration was much more significant than that of both solids and BOD in the beginning of the experimental period. The difference of VFAs levels between the treatment and control ranged from 1600 to 2500 mg/L after only one week aeration. After a month, the VFAs in the aerated manure were all lower than 230 mg/L, and became undetectable after 13 weeks of aeration.

Generally, a total VFAs level of below 230 mg/L corresponds to slurry without offensive odor (Burton, 1992). According to this standard for effectiveness evaluation, a treatment time of around 1 month is needed to reach the steady state for the aeration treatment. According to Figure 4.9, the VFAs reduction efficiency (compared to the initial treatment data) in the first sampling week rapidly reached 68.9% and removal efficiency (compared to the control data) was 41.1%. Since then, the VFAs removal efficiency continued to increase exponentially and reached the range between 89.1% (one month later) and around 95% (three month later).

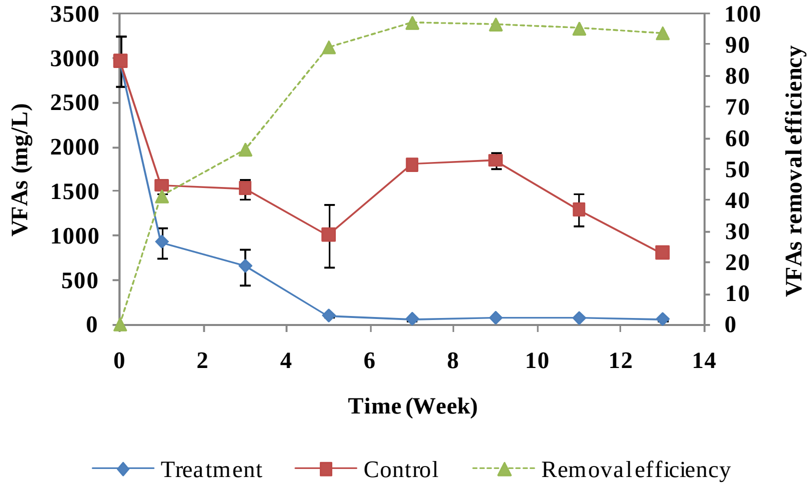


Figure 4. 9 The changes of VFAs and removal efficiency

4.2.3.6 Statistics analysis

Regression analyses among odor generation potential parameters will be helpful to elucidate the stabilization of aerated manure during aeration period as well as the strategy for odor control. The effect of system characteristics (independent variables) on response variables was evaluated using analysis of variance (ANOVA) and regression analysis. ANOVA is similar to regression in that it is used to investigate and model the relationship between a response variable and one or more independent variables. However, it differs from regression in two ways: the independent variables are qualitative (categorical), and no assumption is made about the nature of the relationship. ANOVA was applied by a General Linear Model that required a response or measurement taken from the units sampled as a function of one or more factors. Using the general linear model procedure to conduct an ANOVA analysis assumes that the variances of several samples are equal. The experimental data is shown in Table 4.1 and the correlation coefficients are presented in Table 4.2.

Table 4. 1 The experimental data under surface aeration treatment

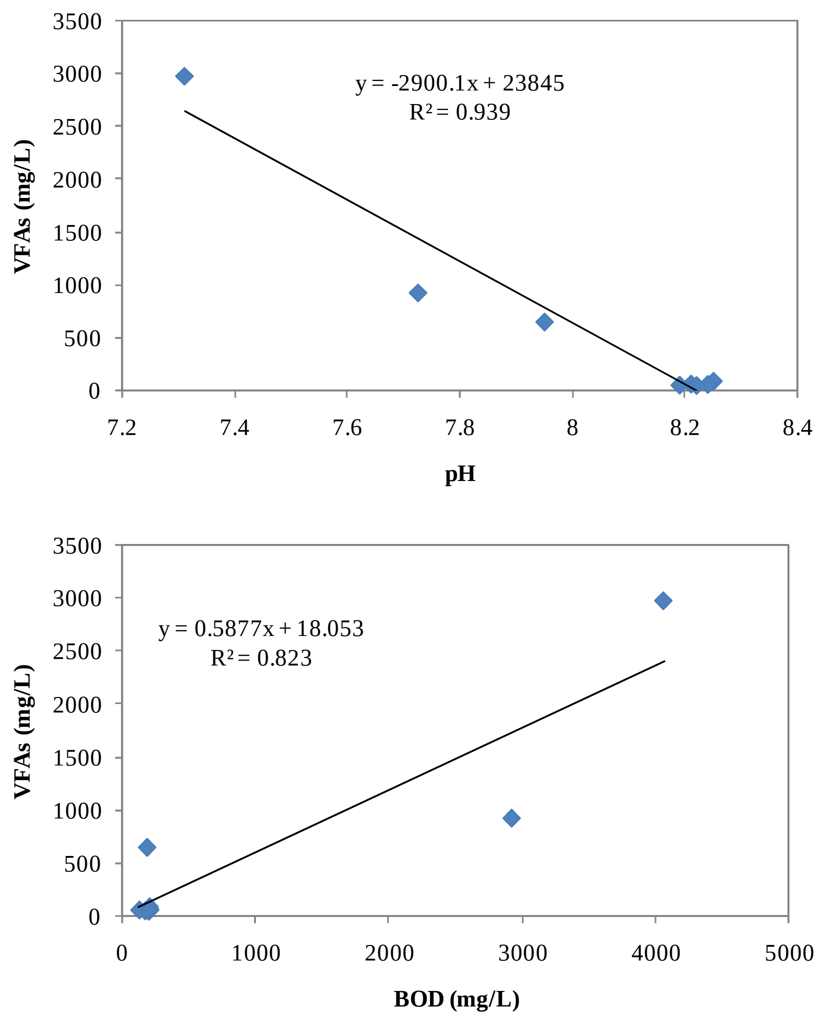
weeks	DO(mg/L)	BOD(mg/L)	VFAs(mg/L)	pH	TS(%)	TVS(%)
0	0.3	4060	2974.85	7.31	0.66	0.37
1	1.57	2920	926.38	7.725	0.67	0.38
3	1.6	180	651.53	7.95	0.55	0.28
5	2.13	200	91.25	8.25	0.49	0.24
7	3.73	166	53.67	8.19	0.44	0.19
9	4.87	206	64.80	8.21	0.45	0.19
11	5.37	124	61.57	8.24	0.42	0.17
13	9.35	194	50.71	8.22	0.36	0.13

Table 4. 2 The correlation coefficients

	DO(mg/L)	BOD(mg/L)	VFAs(mg/L)	pH	TS(%)	TVS(%)
DO(mg/L)	1					
BOD(mg/L)	-0.58	1				
VFAs(mg/L)	-0.62	0.91	1			
pH	0.66	-0.95	-0.97	1		
TS(%)	-0.86	0.85	0.76	-0.87	1	
TVS(%)	-0.86	0.85	0.78	-0.87	0.99	1

From Table 4.2 and 4.3, the correlation coefficients (r) for VFAs versus TVS, BOD, and pH in liquid manure were 0.78, 0.91, and -0.97, respectively. It proved that VFAs was highly significant with TVS, BOD and pH. Previous studies demonstrated similar linear correlations between BOD and VFAs for swine manure with no aerobic treatment (Zhu et al., 2001), and for manure subject to aerobic treatment (Williams, 1984). Research also showed that liquid manure VFAs generation was relevant to the portion of solids mainly in the form of TVS by microbial degradation (Yasuhara et al., 1984;

Zhu et al., 2001). Therefore, although different characteristics on the changes of TVS, BOD, VFAs and pH over time are observed in manure stabilization, the odor generation potential in the aerated manure may still be linearly evaluated by the concentrations of TVS, BOD or pH, with BOD being the best estimator. The effects of pH, TVS and BOD on VFAs were evaluated using ANOVA and regression models and described in Table 4.3. These tables project VFAs values against independent variable. Statistical results for ANOVA are summarized in Table 4.3. The overall linear relationships among VFAs, pH, BOD and TVS are presented in Figure 4.10.



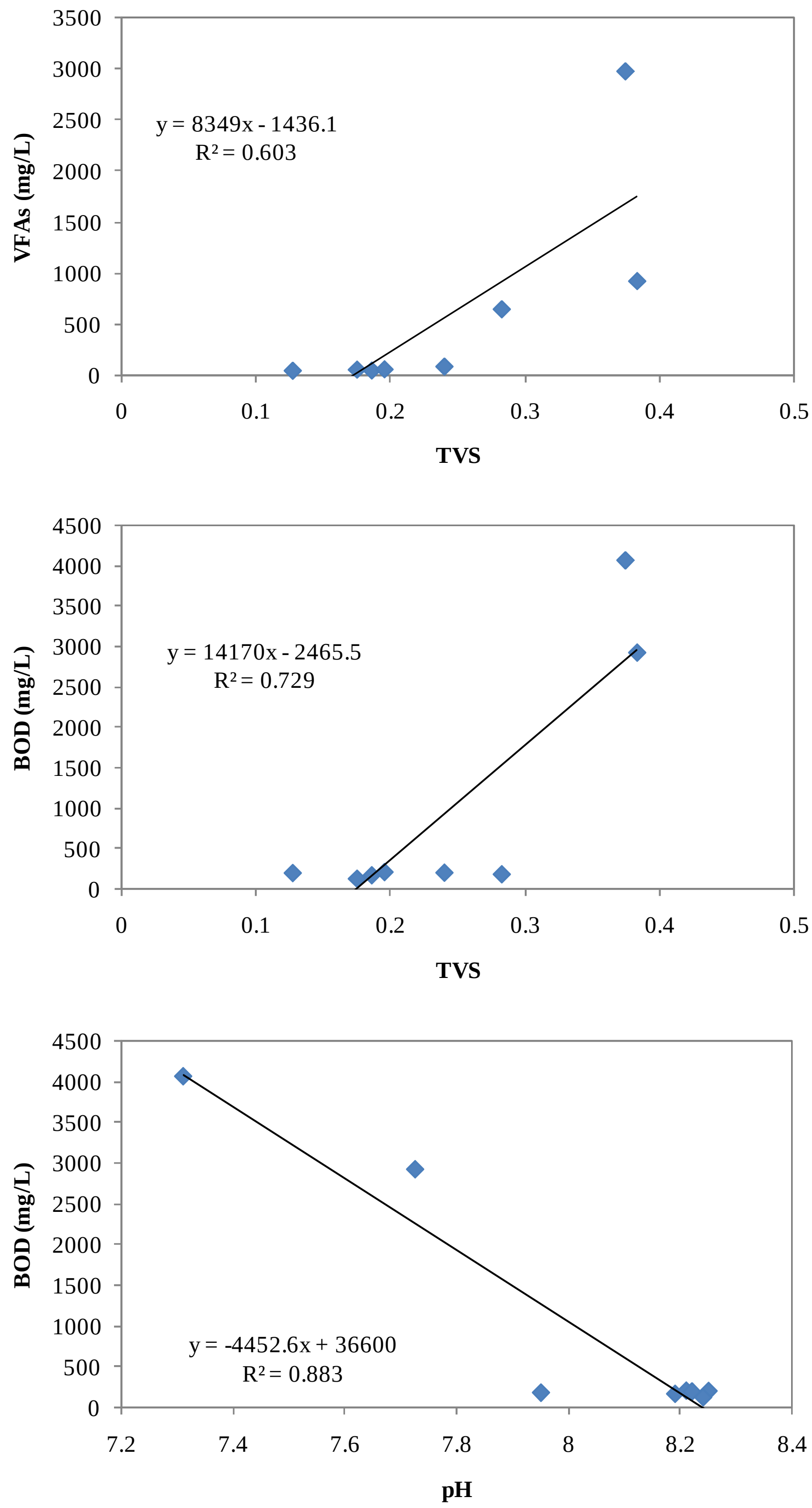


Figure 4. 10 The overall linear relationships among VFAs, pH, BOD and TVS

Table 4. 3 ANOVA Tables

ANOVA **(**Predictors: (Constant), BOD; Dependent Variable: VFA**)**

	Sum of Squares	df	Mean Square	F	p-value
Regression	5908291.672	1	5908291.672	27.793	.002(a)
Residual	1275474.909	6	212579.151		
Total	7183766.581	7			

ANOVA **(**Predictors: (Constant), pH**;** Dependent Variable: VFA**)**

	Sum of Squares	df	Mean Square	F	p-value
Regression	6723784.271	1	6723784.271	87.705	.000(a)
Residual	459982.310	6	76663.718		
Total	7183766.581	7			

ANOVA (Predictors: (Constant), TVS; Dependent Variable: VFA)

	Sum of Squares	df	Mean Square	F	p-value
Regression	4331708.648	1	4331708.648	9.113	.023(a)
Residual	2852057.933	6	475342.989		
Total	7183766.581	7			

The p-value obtained for BOD, pH, and TVS is 0.002, 0.000 and 0.023, respectively. As this value is lower than the level of significance selected (α=0.05), it is concluded that the parameters of BOD, pH, and TVS are significant factors in response to VFAs. The greater the F value, the more importance this parameter reflects. Therefore, a slightly higher F value for the pH factor, however, suggests that this factor is more important in the response of VFAs, than another factor. And also it can be found that BOD is slightly more important in the response of VFAs according to the F value.

Although it is commonly assumed that the BOD in the manure reflects the amount of organic compounds that need to be biodegraded by aerobic bacteria with the help of oxygen (Loughrin et al., 2006), it has to be understood that the BOD is not only composed of VFAs. Thus, a reduction in VFAs may not necessarily show a reduction in BOD at the same level. The relationship of BOD with VFAs from the analysis of all the liquid manure samples in this study is presented in Figure 4.10 and Table 4.3, which features a relatively good linear correlation between these two variables with the correlation coefficient being 0.91. And the coefficient of determination (R^2) obtained in this analysis demonstrated that 82.3% of BOD in the response of VFAs was explained by the General Linear Model.

The value of pH influences the release of VFAs gases and the low initial pH would have promoted release of VFAs gasses due to a greater concentration of the non-dissociated molecules dissolved in the liquid phase. From Figure 4.10, the R^2 obtained in this analysis demonstrated that 93.6% of pH in the response of VFAs was explained by the General Linear Model. The higher value of R^2 confirmed a good model fit with the data obtained. In general, an increase in pH has been previously related to lower VFAs levels (r=-0.97). Thus, for manure collected in this study, the pH value explained most of the observed differences in VFAs values. This means that VFAs should not be contributing significantly to malodour from these manures under higher pH value as the pKa of VFAs is within 4.8-5.0. This was probably due to a combination of an acidity effect and a duration effect.

Volatile solids generally refer to the various organic components with higher biodegradability than the rest of solids that is mainly associated with the coarse material in the slurry and effectively inert in the typical aerobic treatment process (Burton, 1992). This means that the TVS can contribute to the odor generation potential. The relationship of TVS with VFAs from the analysis of all the liquid manure samples in this study is presented in Figure 4.10 and Table 4.3, which features a relatively good linear correlation between these two variables with the correlation coefficient being 0.78. And the coefficient of determination (R^2) obtained in this analysis demonstrated that 60.3% of TVS in the response of VFAs was explained by the General Linear Model. These results are consistent with those from ANOVA table. The parameter of pH is more important in the response of VFAs than another factor in this study and BOD functions better than the parameter of TVS.

Moreover, the change of the ratio of TVS to TS was also considered in this part. The ratio of volatile solids to total solids provides an indication of the efficiency of surface aeration. Increased ratio values indicate that decomposition is less complete and that volatile materials (including odorants) are probably accumulating in the liquor. The relationship between TVS and TS is shown in Figure 4.11 and the ANOVA table is presented in Table 4.4.

From Figure 4.11 and ANOVA table, the coefficient of determination (R^2) obtained in this analysis demonstrated that 99.8% of TS in the response of TVS was explained by the General Linear Model. The higher value of the R^2 confirmed a pretty good model fit with the data obtained. In general, a decrease in TS has been previously related to lower TVS levels (r=0.999). Thus, for manure collected in this study, the TVS value explained most of the observed differences in TS values. This means that TVS should be contributing significantly to TS reduction and then odor generation potential under surface aeration treatment.

Table 4. 4 ANOVA table

	Sum of Squares	df	Mean Square	F	p-value
Regression	.062	1	.062	3321.863	.000
Residual	.000	6	.000		
Total	.062	7			

a Predictors: (Constant), TS; b Dependent Variable: TVS

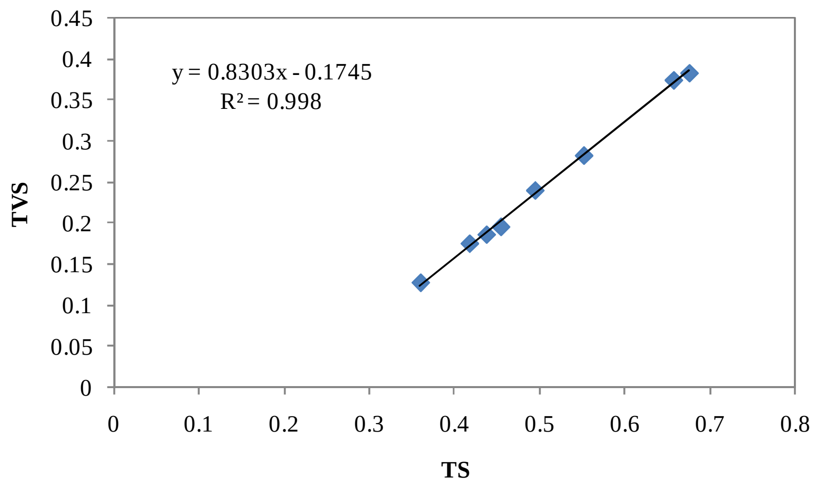


Figure 4. 11 The overall linear relationship between TS and TVS

4.3 Field scale

4.3.1 Materials and methods

4.3.1.1 Experimental setup

The hog farm for the project was located in Nicollet County in Minnesota, which was a growing-finishing facility with 4000 pigs. The buildings were equipped with a pull-plug manure handling system with all wastewater (including wash water) gravity flowing into the outside lagoon for storage, which had a size of roughly one acre (100m x 40m). The schematic of the surface aeration unit used is illustrated in Figure 4.12, which covers only one third of the surface, leaving the uncovered portion as the control. The photos for the actual aeration system are presented in Figure 4.13.

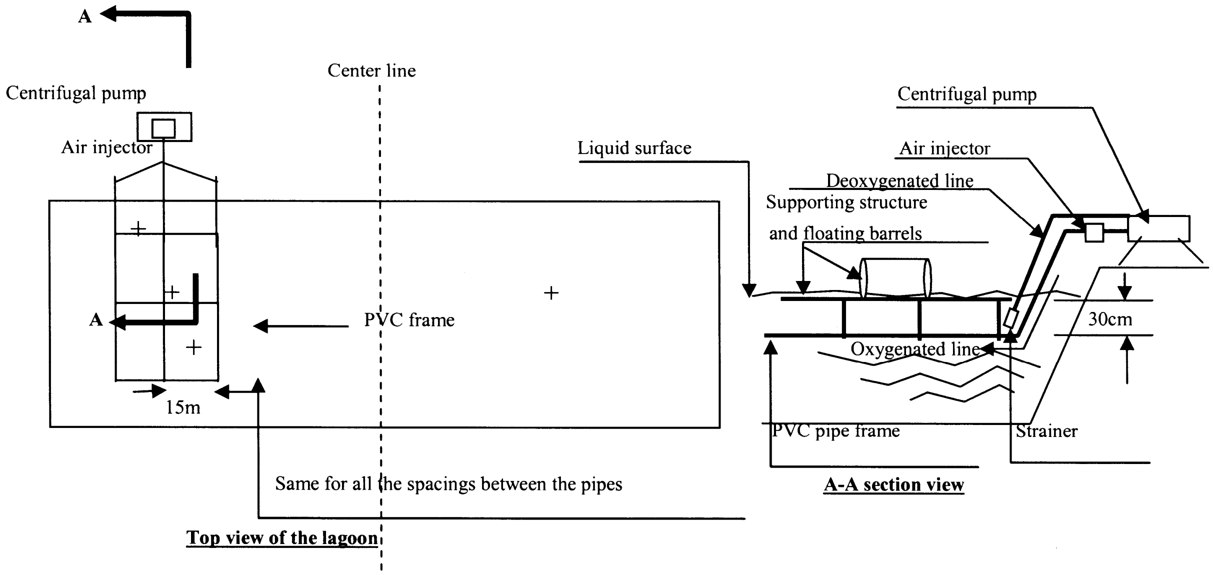


Figure 4. 12 Schematic of the aeration system design (the "+" signs indicate the sampling points)

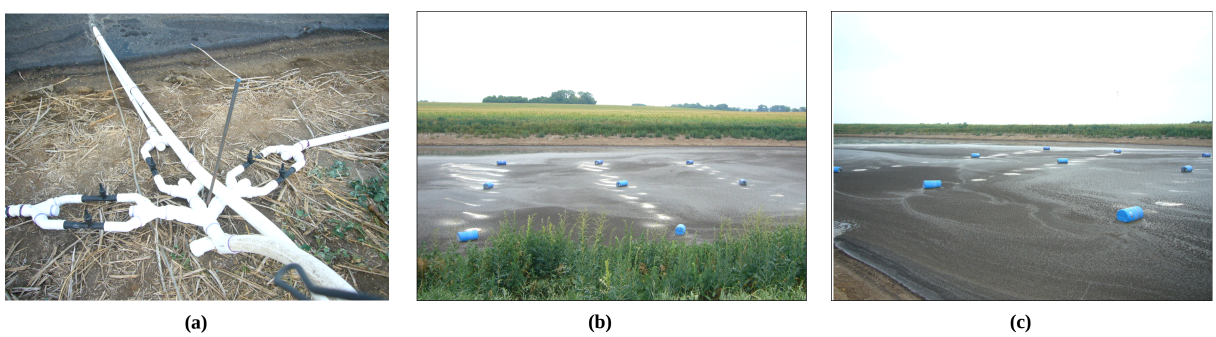


Figure 4. 13 (a) the aeration injector module; (b) and (c) aeration in operation

The aeration system consisted of PVC piping (2.54 cm in diameter) with openings made on the pipes at 1.14 m intervals (facing horizontally for liquid injection) in this study. Pipe size and opening intervals were determined through hydraulic calculations based on the pump capacity, lagoon area, even liquid flow rate from each hole, and flow rate in the pipe. Air was supplied by the venturi air injector module (Figure 4.13(a)) and the entire system was driven by a 1.1 kW centrifugal water pump. The piping structure was placed 30cm below the liquid surface (so the aerated depth was 30cm from the top) and supported by floating barrels (the blue color objects shown in Figure 4.13 (b), (c)) that allowed the structure to float with the liquid level. A strainer was placed in the front of the deoxygenated line to prevent coarse materials from entering the system to clog the aerator. All liquid samples were taken in triplicate 30cm below the surface at locations indicated by a ‘+’ sign in Figure 4.12.

4.3.1.2 Sampling and analysis

Aeration started and ran continuously throughout the entire experimental period, which was about 16 weeks (from mid June to mid October 2008). After the start of aeration, three liquid manure samples (100 mL each) were taken once a week at a depth 30 cm below the liquid surface from the treatment zone and one sample was taken for the control zone (indicated by the “+” sign in Figure 4.12) using automatic samplers (Model 3700, Isco Inc., Lincoln, NE). In addition, at every sampling time, the DO and temperature of the liquid adjacent to the sampling points were measured using an Oakton waterproof DO 300 meter by boat to assess the performance of the system. All liquid samples were analyzed for BOD, VFAs, TS, TVS, and pH following the Standard Methods (AHPA, 1998). Whenever analysis was not done immediately after sampling, the samples were stored at -20°C and then thawed and allowed to reach room temperature prior to analysis. System efficiencies were calculated as the average percentage reduction of water quality indicators in the system with respect to the control concentrations. Significant relationships between water quality parameters and odor indicators were evaluated using linear correlation and regression analysis.

4.3.2 Results and discussion

4.3.2.1 DO levels and pH

Figure 4.14 presents the averaged DO levels measured once a week (every Wednesday morning).

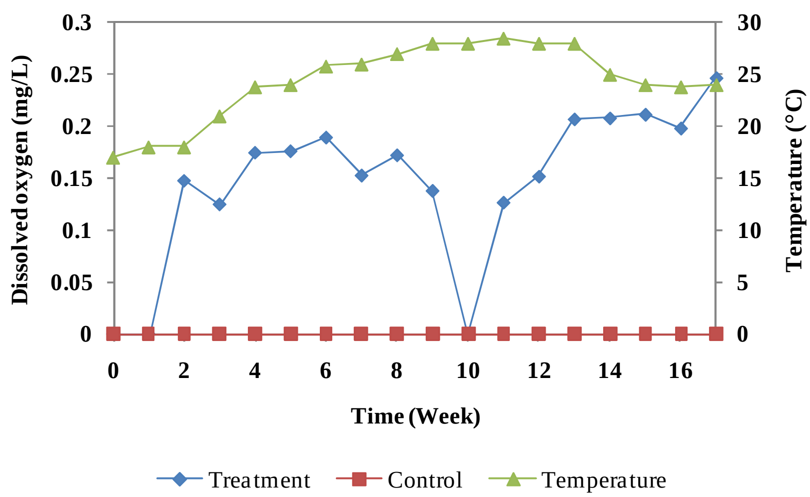


Figure 4. 14 The dynamic changes of DO in liquid manure

According to Figure 4.14, the temperature values ranged from 17 to 28.5 °C and the DO levels in the aerated liquid manure were in general stable ranging from 0.1 to 0.25 mg/L in the course of aeration treatment; while the DO levels from the control were virtually zero. Due to pump failure in the tenth week, the DO in the treatment zone was undetectable during sampling but was recovered and increased after the problem was solved. The fluctuation in the DO levels reflected the difference in the oxygen supply rate and the biological consumption rate (WPCF and ASCE, 1988). The DO concentration increases whenever the oxygen supply exceeds the biological consumption rate, remains constant when the two rates are equal, and decreases when the consumption rate exceeds the supply rate. From Figure 4.14, it can be seen that the oxygen supply exceeded the biological consumption rate after 3 months of operation.

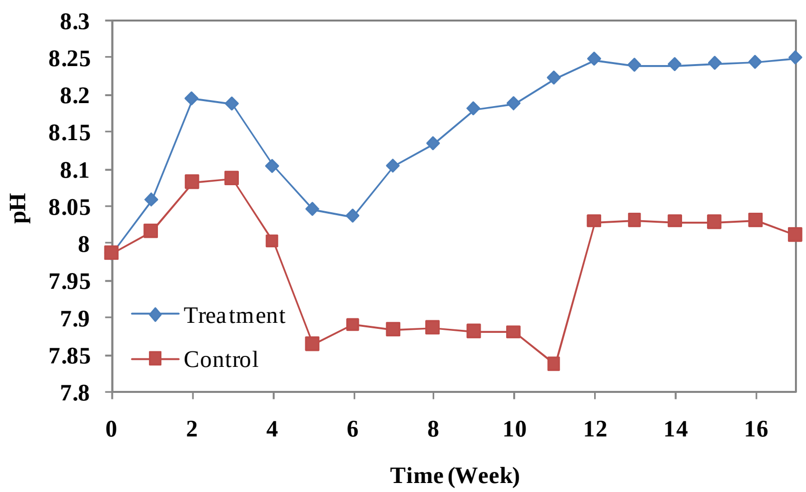


Figure 4. 15 The dynamic changes of pH in liquid manure under treatment and control

The dynamic change in pH during the experiment is presented in Figure 4-15. Surface aeration sharply increased liquid manure pH after one-month continuous operation. From week 5 to 12, pH rose from 8.05 to 8.25, and then maintained around 8.25 for the rest of the test period. An increase in pH was also observed in the control zone. The difference in pH between the treatment and the control ranged from 0.04 to 0.31 with an average of 0.2. The liquid pH increased as the aeration process proceeded, indicating that more organic nitrogen was mineralized to ammonium nitrogen by the aerobic bacteria, supported by the oxygen provided by the aerator. Similar results were also found by Luo et al. (2001) and Zhang et al. (2003).

4.3.2.2 The effect on TS and TVS changes

Table 4.5 shows the changes of TS and TVS levels in the liquid samples under the treatment and the control. It can be seen that the surface aeration system reported herein was not effective to reduce solids content (0.9% after 4 month s aeration treatment compared with 1% no aeration).

Table 4. 5 TS and TVS levels under treatment and control sides (%)

	Treatment			Control*			Removal efficiency	
week	TS	TVS	TVS/TS	TS	TVS	TVS/TS	TS	TVS
1	1.55±0.08	0.76±0.00	49.02±0.02	1.55	0.76	49.02	0.00	0.00
5	1.14±0.02	0.52±0.01	45.72±0.00	1.15	0.53	46.30	0.92	2.16
10	1.01±0.02	0.38±0.02	37.40±0.01	1.02	0.41	40.31	1.13	8.28
12	0.99±0.04	0.40±0.05	40.37±0.04	1.05	0.43	40.88	5.42	6.62
13	0.92±0.05	0.35±0.05	38.57±0.04	1.00	0.41	41.37	8.41	14.59
17	0.90±0.01	0.28±0.04	30.92±0.04	1.07	0.46	42.75	16.55	39.64

*one sample was taken for the control side.

The TVS removal efficiency, which was calculated by dividing the average net reduction (control vs. treatment) by the value of control then multiplied by 100, was generally higher than that of TS (39.64% vs. 16.55% after four months of aeration, respectively). The ratio of TVS to TS decreased from about 49% to 31% in the same treatment period, while no significant difference was found between the treatment and control. Volatile solids generally refer to the various organic components with higher biodegradability than the rest of solids mainly associated with the coarse material in the slurry and effectively inert in the typical aerobic treatment process (Burton, 1992).

Combining the fact of the slightly increasing removal efficiency of TVS with the slight trend of decreasing percentage of TVS/TS has indicated that both the quantity and intensity of the readily degradable solids related to odor generation potential can be reduced by the surface aeration treatment.

4.3.2.3 The effect on BOD changes and the odor generation potential

Figure 4.16 presents data on the fluctuations of BOD concentrations during the experiments. As time went by in aeration operation in the lagoon, a trend that the BOD concentration in the aerated zone slowly decreased was found with at least four distinct features. First, in the first 4 weeks of the experiment, a rising trend for BOD was observed for both treatments with no big difference between them. After that, both BOD levels started to decline till week 9 but the difference between the treatment and control widened (from 450 to 2140 mg/L). Second, the pump failure at week 10 only caused a slight increase in BOD concentration in the liquid and its decline trend resumed immediately after the pump problem was cleared. In the remaining operating period, the BOD level was drastically reduced linearly in the aerated manure. This phenomenon indicates that in the field-scale situation, temporary shutdown of the aeration operation for planned or unplanned reasons, such as equipment failure, maintenance work, electricity outage, etc., will not substantially impair the treatment effect as long as the interruption lasts less than a few days. Third, in the first 7 weeks, the BOD removal efficiency was lower than 20% (about 9% on average). But after 7 weeks of operation, the removal efficiency was increased linearly from 21.5% to 86.5%. The low removal efficiency in the first 7 weeks indicated the minimum ‘break-in’ time needed for the aeration system under study to successfully establish an aerobic bacterial consortium in the top aerated layer. Finally, after 4 months of aeration, the BOD concentration in the aerated liquid was still around 670 mg/L with the removal efficiency lower than 90%. According to Williams (1984), the offensive odor would not become an issue if the BOD concentration in the treated manure was 171 mg/L or lower. Obviously, the system in this study could not achieve that particular BOD concentration in the aerated manure after 4 months of operation.

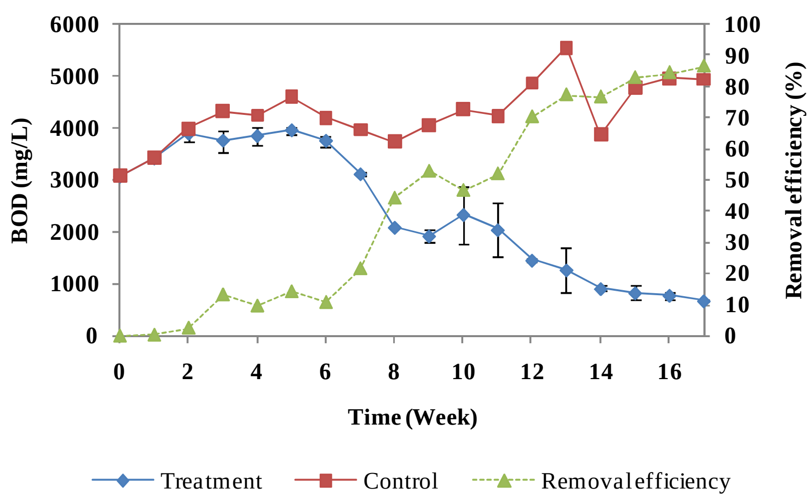


Figure 4. 16 The dynamic changes of BOD in liquid manure

Zhang et al. (2003) found in their lab study that the BOD removal efficiency was stabilized within 90% to 95% after 4 weeks of aeration during which it increased linearly from about 7.5% to 90%. Although the results from Zhang et al. (2003) are not directly comparable to those from this study due to the scale difference, it is reasonable to infer, based on the rising trend of the BOD removal efficiency in Figure 4-16, that the 171 mg/L goal could be achieved by the surface aeration treatment if the operation continues beyond the 4-month period. Therefore, it can be concluded that the field-scale surface aeration system developed and experimented in this study indeed can reduce BOD in the treated liquid manure, and thus the odor generation potential, stored in open ponds and/or lagoons.

4.3.2.4 The effect on VFAs changes and the odor generation potential

The dynamic changes in VFAs shown in Figure 4.17 were similar to that of BOD during the experiments. With the increase in aeration time, a trend that the VFAs levels slowly decreased was found, featuring rapid decreases in the first 2 months (except for the 7th week for unknown reasons) and reduced decreases in the remaining aeration period. Compared to the control, the VFAs removal efficiency increased from 5% to 85% during the operation period, achieving a level of VFAs lower than 230 mg/L in the aerated liquid manure after 4 months of treatment. Generally, a total VFAs level of below 230 mg/L corresponds to slurry without offensive odor (Burton, 1992). By using this number, the aeration time needed for the surface aeration system in this study to achieve inoffensive odor will be around 120 days. In other words, the system has to run continuously for almost 4 months to bring the odor indicator VFAs down to a level that is not offensive.

It appears that the surface aeration module used in this study can reach a DO level of around or above 0.25 mg/L in the lagoon liquid if the operation continues beyond the test period (Figure 4.14). Even so, this is still lower than the minimum level (0.5 mg/L) considered effective in controlling odor by other researchers (Zhang et al., 1997). One obvious reason for low DO lies in the small water pump used (only 1.1 kW) for around 410 m^3 of liquid manure in the surface layer aerated constantly. Doubling the pump size should certainly increase the DO level, but, at the same time, the running cost as well, which is undesired. That being said, considering a clear trend of VFAs and BOD reduction over the treatment period and the ≤ 5 hp per acre power consumption limit as a rule of thumb, there is no doubt that the surface aeration system presented herein with the current pump size is capable of alleviating odor generation even at DO concentrations lower than 0.5 mg/L after 4 months of continuous operation. Therefore, in areas where surface aeration is permitted by weather to run all year around, the system developed and field evaluated in this project can be an inexpensive but effective technique to control odor emissions from open manure storages after 4 months of start-up period.

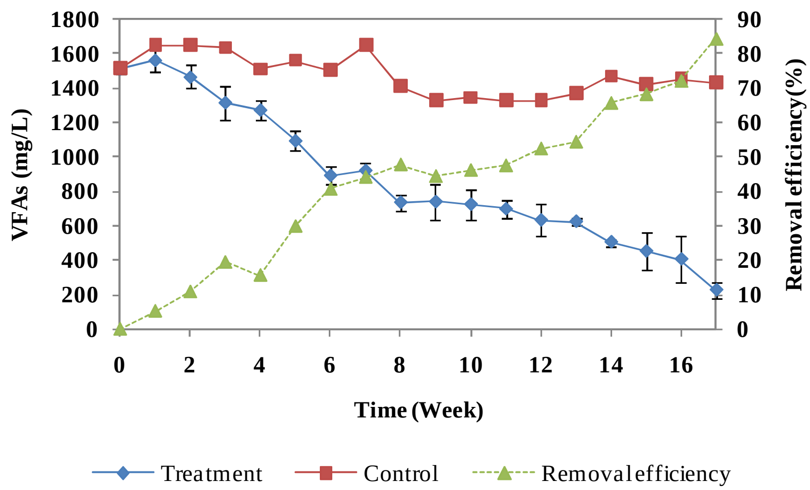


Figure 4. 17 The dynamic changes of VFAs in liquid manure

4.3.2.5 The effect of surface aeration on COD

Figure 4.18 presents data on the fluctuations of COD concentrations during the experiments. As time went by in aeration operation in the lagoon, a trend that the COD concentration in the aerated zone very slowly decreased was found except for the 4^{th}

month. Total carbon (i. e., COD) removal will occur by three processes: (i) utilization of COD for biological denitrification which is actually reduction of nitrate and oxidation of carbon (oxygen source-nitrate); (ii) COD removal by aerobic oxidation (i. e., by oxygen transferred from air); (iii) carbon converted to biomass (Dhamole et al., 2009). From Figure 4.18, it can be found that the surface aeration in this study has insignificant effect to COD removal.

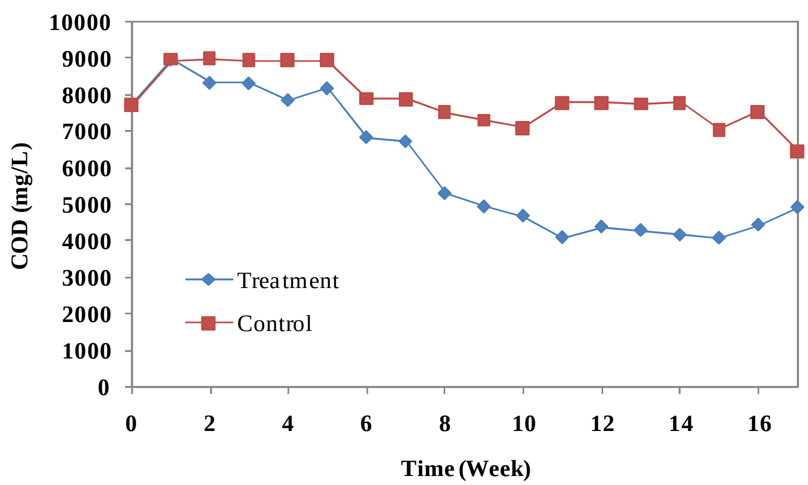
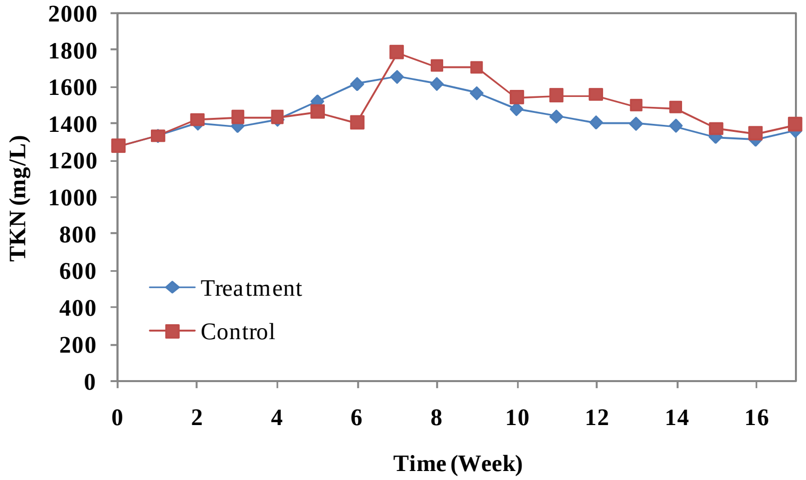


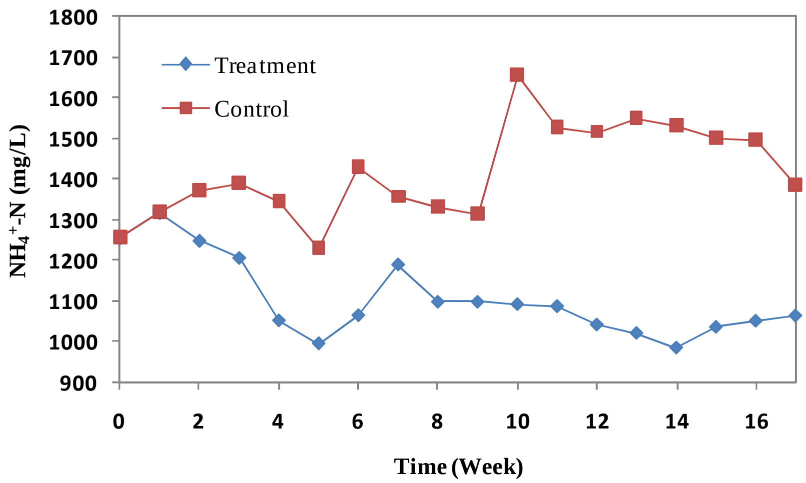
Figure 4. 18 The dynamic changes of COD in liquid manure

4.3.2.6 The effect of surface aeration on nitrogen components

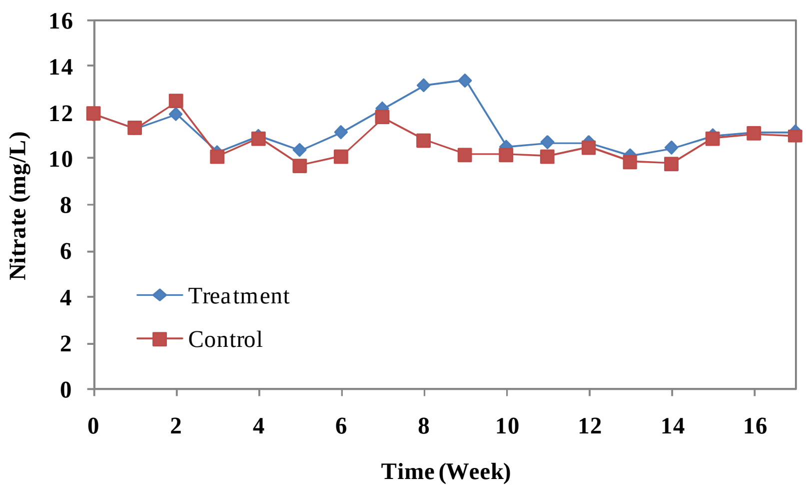
Figure 4.19 presents data on the fluctuations of nitrogenous components concentrations during the experiments. Under aerobic conditions, the nitrogen compounds (proteins, peptides, and amino acids) are converted to ammonium (NH_4^+) by heterotrophic bacteria (require nourishment from organic substances) and then oxidized by autotrophic bacteria (obtain nourishment from inorganic matter such as (NH_4^+) to nitrite (NO_2^-) and then to nitrate (NO_3^-). The degree of oxidation depends on the amount of oxygen provided and the reaction time allowed in the treatment process.



TKN (mg/L)
Time (Week)
Treatment
Control



NH_4^+-N (mg/L)
Time (Week)
Treatment
Control



Nitrate (mg/L)
Time (Week)
Treatment
Control

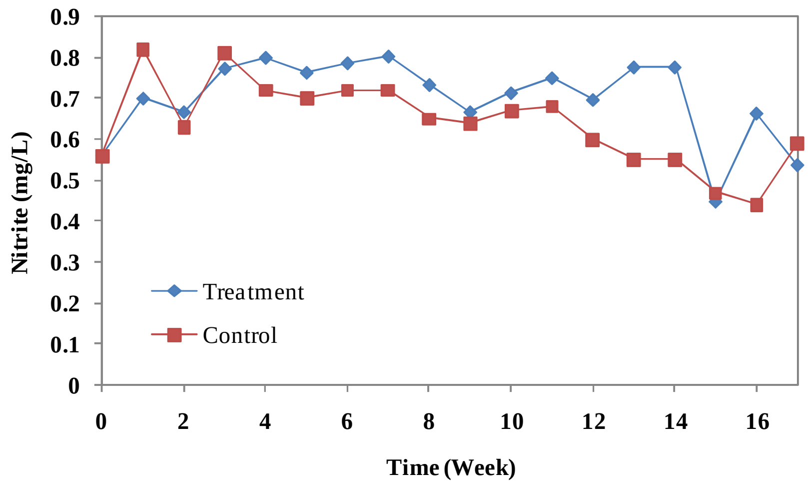


Figure 4. 19 The dynamic changes of nitrogen components in liquid manure

From Figure 4.19, it can be found that the surface aeration in this study has no effect on the nitrogen components removal. The reason is that aeration at this low rate usually cannot efficiently convert ammonia to nitrate, thus likely causing high aerial emissions of ammonia. In this study, the ammonium in the liquid is mostly constant under the surface aeration treatment and just a little lower compared to the controlled side. Unfortunately, there is no information to explain this from this study. Form Figure 4.18 and 4.19, it implied that under surface aeration (low rate), slightly reduction of COD and no effect on nitrogen components were found in this study. Yamagiwa et al. (1995) found the similar results showing that with free cells, 90% total organic carbon efficiency was achieved with only 50% nitrogen removal.

4.3.2.7 The correlation between the two indicators - BOD and VFAs

Comparing the VFAs and BOD data from this study reveals that the aeration requirement for reducing BOD for odor abatement during manure storage is more stringent than for reducing VFAs because the treatment is not able to whittle BOD to the level perceived to be desirable in terms of offensive odor prevention (171 mg/L) in the experimental duration. Although it is commonly assumed that the BOD in the manure reflects the amount of organic compounds that need to be biodegraded by aerobic bacteria with the help of oxygen (Loughrin et al., 2006), it has to be understood that the BOD is not only composed of VFAs. Thus, a reduction in VFAs may not necessarily show a reduction in BOD at the same level. The relationship of BOD with VFAs from the analysis of all the liquid manure samples in this study is presented in Figure 4.18, which features a relatively good linear correlation between these two variables with the correlation coefficient being 0.87.

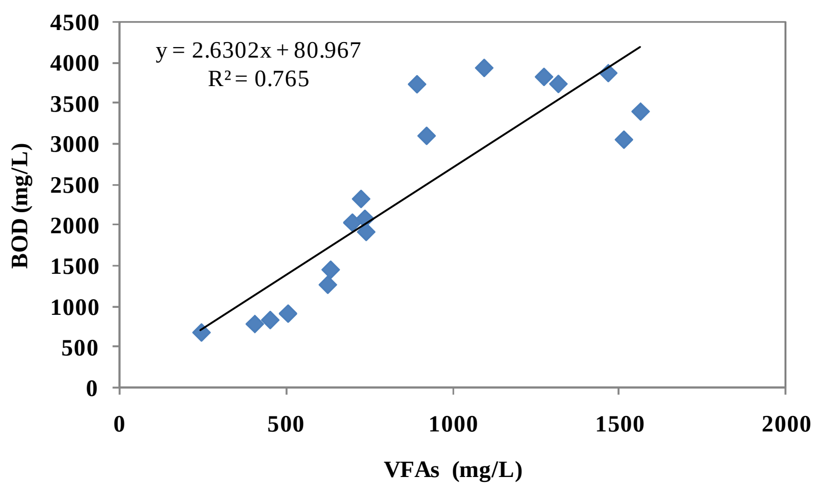


Figure 4. 20 The correlation between the BOD and VFAs concentration

Also seen in Figure 4.20 is the determination of correlation which is around 0.76, indicating that about 76% of BOD can be explained by the VFAs concentration in the manure. Therefore, based on the data from this study, it may be inferred that the VFAs in manure could be used as a more appropriate indicator than BOD in determining the effectiveness of aeration treatment for odor reduction because the changes in VFAs caused by the aeration treatment will directly affect the odor generation potential but may not be reflected in the changes of BOD concentration.

The effects of VFAs on BOD were evaluated using ANOVA and regression models and described in Table 4.5. The p-value obtained is 0.000. As this value is lower than the level of significance selected (α=0.05), it is concluded that the parameters of VFAs is a significant factor in the response of BOD. Compared to the results from lab-scale study, the relationship between BOD and VFAs is slightly less obvious.

Table 4. 6 The ANOVA Table for BOD and VFAs

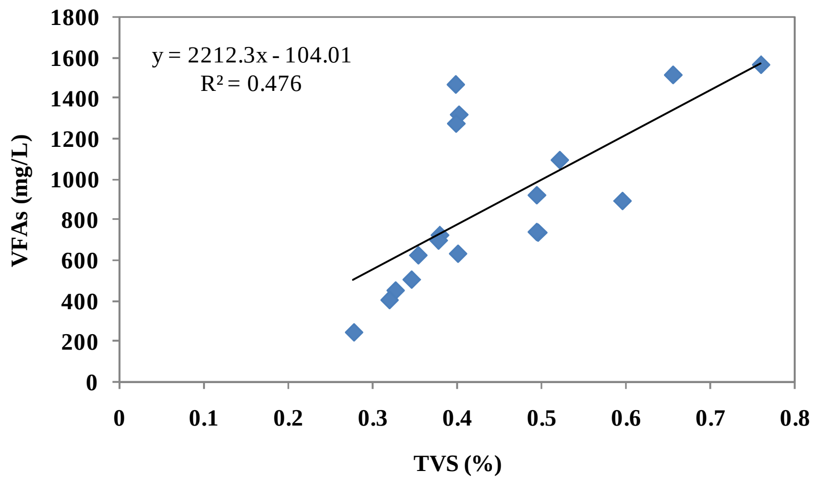
	Sum of Squares	df	Mean Square	F	p-value
Regression	19304367.325	1	19304367.325	51.967	.000(a)
Residual	5943621.400	16	371476.337		
Total	25247988.725	17			

4.3.2.8 The correlation between VFAs and pH, TVS

The correlations between VFAs and pH, TVS are presented in Figure 4.19. The effect of system characteristics (independent variables) on response variables was evaluated using ANOVA table and regression analysis (Table 4.6). From Figure 4.19 and Table 4.6, the correlation coefficients (r) for VFAs versus TVS and pH in liquid manure were 0.69 and -0.71, respectively. It proved that VFAs was also found highly significant with TVS and pH at field-scale study.

Compared to the results from lab-scale study, the parameter of VFAs is not well related to pH at field-scale experiments. From Figure 4.10, the R^2 obtained in this analysis demonstrated that only half of pH in the response of VFAs was explained by the

General Linear Model. However, in general, an increase in pH has still been previously related to lower VFAs levels (r=-0.71, and r is -0.97 for lab-scale study). This means at field-scale surface aeration experiments, pH value can't be used to explain most of the observed differences in VFAs values as there are some environmental parameters that can influence the dynamic changes of observed parameters (i. e., weather, loading et al.).



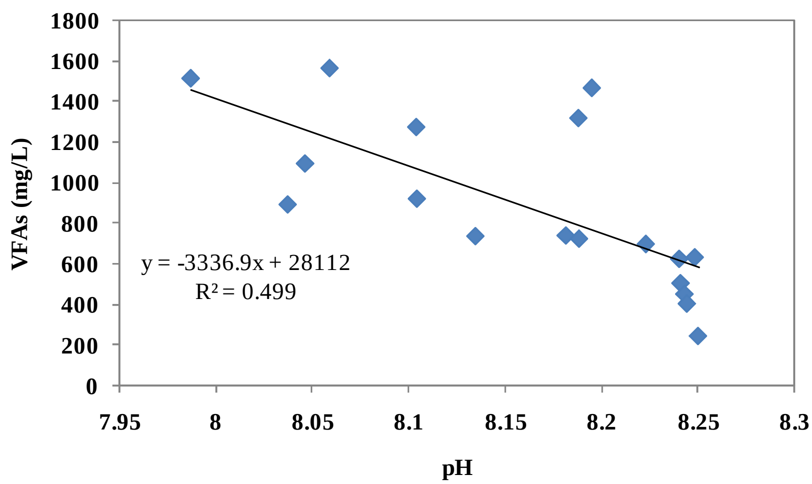


Figure 4. 21 The correlation between VFAs and pH, TVS concentrations

The same applies to pH, i.e., the relationship of TVS with VFAs is also worse than that from lab-scale experiments, although the TVS is less related than pH and BOD to VFAs at lab-scale. From Figure 4.10, the R^2 obtained in this analysis demonstrated that only 47.6% of TVS in the response of VFAs was explained by the General Linear Model. However, in general, a decrease in pH has still been previously related to lower VFAs levels (r=0.69, and r is 0.78 for lab-scale study). This means that the value of TVS was found to contribute to the odor generation potential.

Table 4. 7 ANOVA Tables

ANOVA (a Predictors: (Constant), pH; b Dependent Variable: VFA)

	Sum of Squares	df	Mean Square	F	p-value
Regression	1392498.805	1	1392498.805	15.937	.001(a)
Residual	1398015.078	16	87375.942		
Total	2790513.883	17			

ANOVA (a Predictors: (Constant), TVS; b Dependent Variable: VFA)

	Sum of Squares	df	Mean Square	F	p-value
Regression	1328362.546	1	1328362.546	14.536	.002(a)
Residual	1462151.337	16	91384.459		
Total	2790513.883	17			

The p-value obtained for pH and TVS is 0.001 and 0.002, respectively. As this value is lower than the level of significance selected (α=0.05), it is concluded that the parameters of pH and TVS are significant factors in the response of VFAs. The much greater F value reflects the more importance of this parameter. The F values obtained for pH and TVS are almost the same, implying that pH and TVS play the same important role in the response of VFAs according to the F value. However, compared to the results from Table 4.5, a much higher F value for the BOD factor suggests that this factor is more important in the response of VFAs than another factor at field-scale. This is different from the results at lab-scale.

4.3.2.9 Running cost estimates for this system on a per pig produced basis

Considering the current aeration scheme with three 1.1 kW pumps for a one-acre manure lagoon serving 4000-head finishing pig spaces, the energy cost per pig space, thus per pig produced, can be estimated if the electricity price is $0.07/kWh. Given that the aeration unit will be run all year around, the total electricity cost will be 3 x 1.5 hp x 0.75 kW/hp x 24 hrs/day x 365 days x $0.07/kWh = $2,069.55. Assuming 2.5 cycles of production per year, the cost per pig produced will thus be $2,069.55/10,000 pigs = $0.21/pig.

There are three final comments on the surface aeration system investigated in this project. First, it has to be recognized that the existing aeration capacity is insufficient in controlling odor in the first four months of operation, as shown by the data presented in the previous sections. For producers who like the surface aeration system for its simplicity and low-cost features but want to achieve a faster response, one strategy that may be adopted to shorten the "break-in" period is to start the aeration operation with a larger pump (e.g., using a 2 kW pump instead of 1.1 kW) to quickly bring down the levels of odorous compounds in the surface water. After that, the system will return to its original configuration run by the smaller pump. Since most of these pumps have the same size fittings at inlets and outlets, switching pumps for the aeration system is relatively easy to accomplish and will not cause significant rebounds in BOD and VFAs during the downtime, as indicated earlier (Figure 4.16 and 4.17).

Second, from this field-scale study, the total volume (or water level) in the lagoon is a very important factor to influence the surface aeration efficiency. Changing the total volume causes an unstable surface aerated layer and reduces the removal efficiency of odor generation potential in the top aerated layer. Therefore, a stable volume in the lagoon is recommended to obtain better surface aeration efficiency. Last, the surface aeration unit used in this study is limited by the weather. Potential "freeze-up" of piping structure may occur during winter months, especially in a cold climate with freezing temperatures. In addition, the organic matter in the lagoon is biodegraded by various microorganisms that are subject to fluctuations in manure temperature, so winter removal performance by surface aeration may be reduced. However, quantitative field evaluation data are needed to verify and support this strategy.

4.4 Conclusions

Results obtained in this study showed that using the surface aeration system with venturi injectors can efficiently reduce odor generation at both lab and field scales.

4.4.1 Lab scale

The intermittent surface aeration system is able to increase the oxygen concentration in the manure. The aeration treatment increased the solids removal efficiencies from 9.26% to 23.20% for TS and 15.53% to 45.78% for TVS, and decreased the ratio of TVS to TS from 0.57 to 0.35. The BOD removal efficiency remained stable around 90% to 95% after 3 to 4 weeks of aeration during which it increased linearly from about 7.5% to 90%. The VFAs in the aerated manure became nearly undetectable after 13 weeks of aeration. The VFAs removal efficiency increased from 68.90% after one week to 89.1% (one month later) and around 95% (three month later). According to the results and references, a treatment time of 3 to 4 weeks will be needed to stabilize the liquid manure in order to maintain the VFAs level below 230 mg/L.

The correlation coefficients (r) for VFAs versus TVS, BOD, and pH in liquid manure were 0.78, 0.91, and -0.97, respectively. It proved that VFAs was highly significant with TVS, BOD and pH. The p-value obtained for BOD, pH, and TVS is 0.002, 0.000, and 0.023, respectively. As this value is lower than the level of significance selected (α=0.05), it is concluded that the parameters of BOD, pH, and TVS are significant factors in response to VFAs. The much greater F value reflects the more importance of this parameter. Therefore, a slightly higher F value for the pH factor suggests that this factor is more important in the response of VFAs, than another factor. And also it can be found that BOD is slightly more important in the response of VFAs according to the F value.

4.4.2 Field scale

Data showed that the surface aeration system studied could achieve removal efficiencies of 39.64% and 16.55% for TVS and TS, respectively, after 4 months of continuous operation. The ratio of TVS to TS decreased from about 49% at the beginning to 31% at the end of the experimental period, while no significant difference was found between the treatment and the control. In the first 7 weeks, the BOD removal efficiency was lower than 20% (9% on average) and, afterwards, increased linearly from 21.5% to 86.5%. In contrast, the VFAs removal efficiency steadily increased from 5% to 85%

and the levels of VFAs in the aerated liquid manure reached < 230 mg/L after 4 months of treatment. A linear relationship was found between BOD and VFAs with a correlation coefficient of 0.87 and the determination of correlation of around 0.76.

The correlation coefficients (r) for VFAs versus TVS and pH in liquid manure were 0.69 and -0.71, respectively. It proved that VFAs was also highly significant with TVS and pH at field-scale study. The p-value obtained for pH, and TVS is 0.001 and 0.002, respectively. As this value is lower than the level of significance selected (α=0.05), it is concluded that the parameters of pH and TVS are significant factors in the response of VFAs. The F values obtained for pH and TVS are almost the same, implying that pH and TVS play the same important role in the response of VFAs according to the F value. A much higher F value for the BOD factor, however, suggests that this factor is more important in the response of VFAs than another factor at field-scale. This is different from the results at lab-scale.

The information obtained from this study indicates that aeration at DO concentrations lower than 0.5 mg/L can also achieve odor reduction, but the aeration time to reach the satisfactory levels of odor indicator compounds, i.e., BOD and VFAs, may not be applicable in all areas, especially those having short summer windows. This shortfall may be overcome by using a larger pump at the beginning of the operation; however, more research and field evaluation are needed to quantitatively determine the advantages of such an endeavor.

A rough estimate shows that the electricity cost for running this particular surface aeration system is around $0.21 per pig finished.

Chapter 5 Effect of aerator module designs, liquid flow rate, surface aeration depth and alpha factor on aeration efficiency

5.1 Introduction

Oxygen transfer, the process by which oxygen is transferred from the gaseous to the liquid phase, is a vital part of a number of wastewater treatment processes and/or odor controlling processes. The functioning of aerobic processes depends on the availability of oxygen and the rate of oxygen transfer. Sufficient oxygen to meet the requirements of aerobic waste treatment does not enter water through normal surface air-water interface. To transfer the large quantities of oxygen that are needed, additional interfaces must be formed. Oxygen can be supplied by means of air or pure oxygen bubbles introduced to the water to create additional gas-water interfaces. The diffused or bubble aeration process consists of contacting gas bubbles for the purpose of transferring gas to the water. For good performance the rate of supply of dissolved oxygen should be equal to the rate of oxygen consumption exerted by the mixed liquor under any given set of circumstances.

Based on the oxygen transfer efficiencies, research has indicated promising results in using aeration to control odor in lagoons (Pain et al., 1990; Sneath et al., 1992; Zhang et al., 2004; Zhang and Zhu, 2005), with surface aeration being identified as the least costly thus having greater potential to become a means for effective odor control for open manure storage (Zhang et al., 1997; Zhu et al., 2007). There are different formats that aeration is applied in the field, among which using venturi principle to design aeration apparatuses has become popular in recent years, as reported by many researchers (Baylar and Emiroglu, 2003; Bagature, 2005; Baylar and Ozkan, 2006). The results showed that venturi had high air injection efficiencies that could ensure effective aeration in wastewater treatment systems.

Aeration transfers oxygen into a liquid medium by either diffusing gas through a gas-liquid interface, or dissolving gas into the liquid solution. Therefore, oxygen transfer performances can be affected by air flow rate, diffuser submergence, contaminant, and horizontal flow velocity (Vogelaar et al., 2000; Capela et al., 2002; Gillot et al., 2005). These parameters determine factors such as bubble size and the degree of turbulence. Plenty of work in this area has been conducted (Gillot et al., 2000; Hwang and Strenstrom, 1985; Fonade, 2001) showing that in clean water, any increase in the air flow rate would result in a reduction of the residence time of the air bubbles in the water, and consequently of the oxygen transfer efficiency. Under process conditions, an increase in the air flow rate also results in a decrease of the oxygen transfer efficiency. Gillot et al. (2005) found that the increase in the horizontal velocity also induced an increase in the oxygen transfer.

However, it did not always match that observed in clean water, leading to a variation in the alpha factor (ratio of the oxygen transfer coefficients in process to clean water at equivalent conditions of temperature, mixing, geometry). Besides, the diffuser submergence (also called surface aeration depth) can influence the aeration efficiency. The effect of increase in the aeration depth can be explained by the mechanism of exchange of the oxygen from the gas to liquid. During this phase, the transfer is occurring by processes of bubble formation, release, and ascension. This rate of oxygen transfer is dependent on the relative rate of ascent, bubble size, partial pressure of oxygen, temperature, and driving force (the difference between the liquid-film oxygen concentrations in equilibrium with the gas bubble and the bulk-liquid dissolved oxygen content) (Vogelaar et al., 2000; Capela et al., 2002; Gillot et al., 2005).

Moreover, as a surface aerated lagoon generally has large volumes, the liquid (effluent) flow rates, determined by the flow rate meter connected in series to the pump, are considered in order to reach the residence time needed to obtain the required degradation performance. The liquid flow rate is related to the cost, the air flow rate, and horizontal flow rate in the surface aeration system with venturi injectors. This is important to make aeration a truly cost effective method for animal producers to control odor from lagoons. Therefore, there is an acute need to understand the influence of liquid flow rate on oxygen transfer in a surface aerated lagoon using venturi injectors.

In summary, the liquid flow rate, surface aeration depth and contaminants are thus the three governing factors of aeration efficiency in the surface aeration system with venturi injectors. Besides, the effect of aerator module designs under varying liquid flow rates on aeration efficiency was also considered in this Chapter.

The aim of the work presented was therefore to study the influence of aerator module design, the liquid flow rate, alpha factor and surface aeration depth on the oxygen transfer efficiency in a venturi surface aeration system.

5.2 Materials and methods

5.2.1 Aerator modules design

Three air injection modules consisting of two and three venturi air injectors were studied in this chapter to discuss the effect of liquid flow rate, contaminants, and surface aeration depth on aeration efficiency. Zhu et al. (2007) found that a parallel design achieved better oxygen transfer coefficient and could be operated by a smaller pump than a series design. Similar results were obtained and presented in Chapter 3 of this thesis. Therefore, in order to get more information about the influence of the module design, contaminants, surface aeration depth and liquid flow rate on oxygen transfer efficiency, the parallel designs with two (module a, Figure 5.1) and three aerators (module b and c, Figure 5.1) were constructed and experimented in this part. For the parallel design, the transfer of oxygen was perceived to be evenly shared by all air injectors in the aerator module, with each injector only responsible for aerating a portion of the total liquid volume. The reduced liquid volume flowing through each individual air injector allowed the use of smaller injectors in order to maintain the needed liquid flow rate for good air entrapment. All

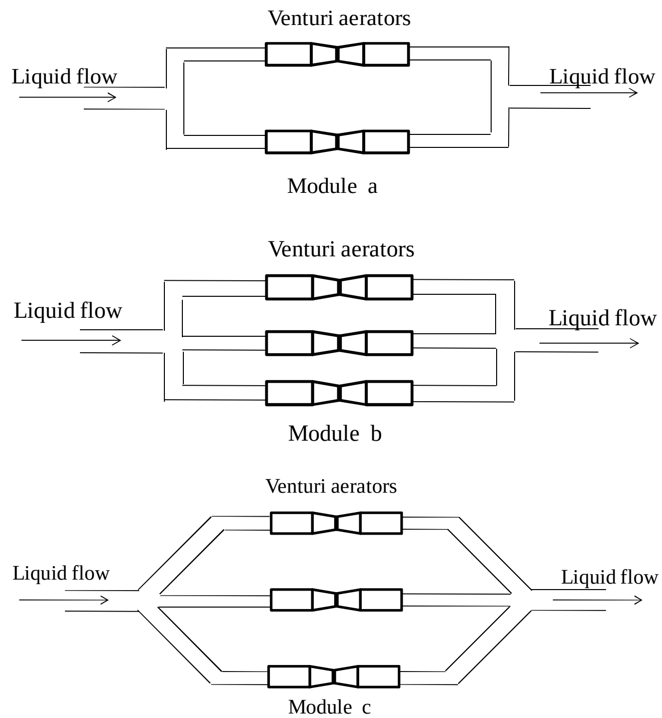


Figure 5. 1 Schematic of the three parallel modules design

venturi aerators used in this project were purchased from the Mazzei Injector Corporation (model#: 1583; thread diameter: 3.8 cm; 500 Rooster Drive, Bakersfield, CA 93307). According to the information from the injector vendor (Mazzei Injector Corp., Bakersfield, Calif.), the particular model used in this study requires a liquid flow rate of 1.2 L/s to obtain the best air suction capability. In our design, the total maximal liquid flow rate that can be provided by the pump is 3.6 L/s, indicating that maximal three air injectors could be connected in parallel in order to provide a sufficient liquid flow rate for each air injector, assuming even distribution of liquid among the three injectors. The injectors in the parallel design were placed 20 cm apart from each other. All venturi air injectors used for different designs were the same model and size and connected by pipes 3.8 cm in diameter (model no. 1583; inlet and outlet internal diameters: 2.9 cm; thread diameter: 3.8 cm; injector length: 28 cm; nozzle diameter: 0.8 cm; Bakersfield, Calif.).

5.2.2 Experimental setup

Two laboratory scale tanks (4.55 m^3 and 9.46 m^3) were used to carry out the experiments reported herein. A water tank with a volume of 4.55 m^3 was used for the study on the effect of aerator module designs, liquid flow rate and alpha factor on aeration efficiency while the other with a volume of 9.46 m^3 was used for the study on the surface aeration depth on oxygen transfer efficiency.

The schematic of the apparatus for determining aeration efficiency under different conditions is shown in Figure 5.2 (heating lamp is used for surface aeration depth study). Both tanks were equipped with a pump, a data acquisition system, dissolved oxygen probes, and piping materials. The probes were calibrated immediately before and after each run. When operating, the centrifugal water pump circulated the liquid through the venturi air injection module to entrain air into the liquid. The detailed program for clean water test with the 4.55 m^3 tank was presented in Chapter 3.

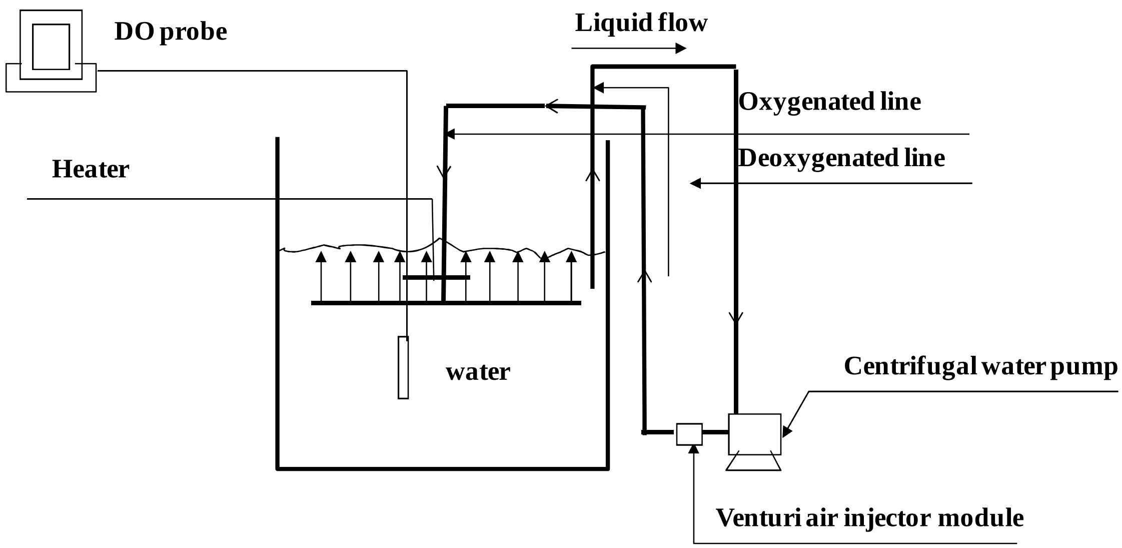


Figure 5. 2 Schematic of the apparatus for determining aeration efficiency of the aerators

The aeration experiments in this part were performed in two phases using tap water (clean water) and separated liquid swine manure (process water) using the smaller tank (4.55 m^3). Clean water test was used to analyze the effect of liquid flow rate, and aerator module designs while process water test was for studying the alpha factor with respect to different effluent liquid flow rate. The liquid flow rate was measured by the flow rate meter connected in series with the pump. For each test, roughly 2.28 m^3 of water or process water was used for determining oxygen transfer coefficients.

For the study on the effect of surface aeration depth on aeration efficiency using the 9.46 m^3 tank, the aeration piping systems were placed at 100, 200, 300, 400 and 500 mm down from the water surface. DO profiles were measured and recorded at the top of the surface aerated layer at each location of aeration piping systems. Compared with the smaller tank, a 0.5 hp pump was used to achieve low flow velocity in the piping systems and avoid liquid turbulence in the tank.

5.2.3 Oxygen transfer function, non steady state method

The apparatus presented in Figure 5.2 was used for the experiments to determine the optimal aeration efficiency for different modules and liquid flow rates used in the system. The testing programs for the venturi surface aeration systems were performed in both clean and process water situations (for alpha factor). The DO inventory of the bulk liquid is monitored during reaeration by measuring DO concentrations at 1-minute intervals until it approached saturation. The test was repeated for three times and the mean values for all measurements were presented. In the clean water test, the water was deoxygenated by adding sodium sulfite (150 mg/L) and cobalt chloride (1 mg/L) before aeration. In addition to DO measurements, water temperatures were also recorded for each individual run using a thermometer located in the center of the water tank. Using the measured DO profile and temperature, a non-linear regression analysis was performed to find the volumetric oxygen transfer coefficient in clean water (k_La) and process water ($k_La\ _p$) that provided the best fit to the experimental data. A detailed program was presented in Chapter 3. The basic theory and program for process water test is similar to the clean water test. Detailed experimental test procedure and techniques used were primarily specified by the ASCE Standard (ASCE, 1992, 1996).

The alpha factor (α), which is the ratio of the oxygen transfer coefficient under process conditions (k_La_p) to the oxygen transfer coefficient in clean water (k_La), was calculated using

$$\alpha = \frac{k_L a_p}{k_L a} \qquad (1)$$

5.3 Results and discussions

5.3.1 The effect of aerator module designs

5.3.1.1 Variation of dissolved oxygen with time for different aerator modules

Data from three runs of each of the three aerator modules (described in Figure 5.1) in the clean water test were averaged and presented in Figure 5.3, which depicted the changes in DO concentration during aeration.

It can be noticed that the dynamic changes on DO concentration vs time with different aerator module designs under varied effluent liquid flow rates were identical in the system. Aerator module a with two air injectors gave a better oxygen transfer capability than module b, and however, for module c, with the reduced friction and a more even flow rate in each line, it was the most efficient in terms of oxygen transfer among all the conditions tested. And the results from this part were similar to those in Chapter 3. However, it was found that the DO difference was varied under different effluent liquid

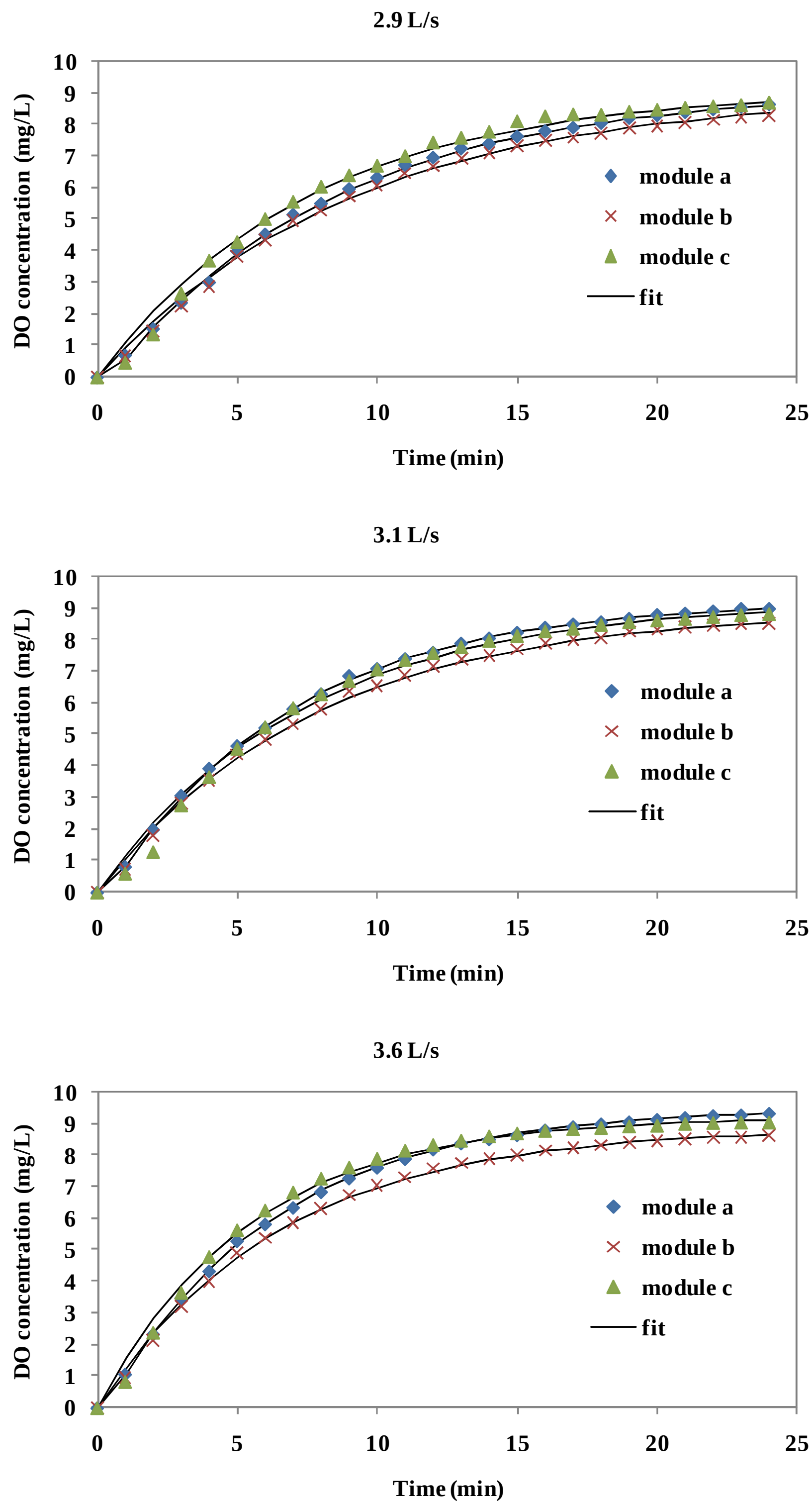


Figure 5. 3 Typical non steady state reaeration curves with different aerator module designs

flow rates. It implied that the effect of friction on liquid flow rates was different. The fundamental goal of efficient aeration is to maximize the dissolution of oxygen into liquid from gaseous phase. On that account, module c is the best among three effluent liquid flow rates investigated. Module b with three aerators didn't work well and entrapped less air than module c due to friction difference.

5.3.1.2 Effect of aerator module designs on aeration efficiency

A semi-log graph based on Equation 2 that reflects the most important parameter in evaluating any aeration system in terms of efficiency was presented in Figure 5.4, in which the slope of each linear regression line is the k_La for that particular aerator module. Since a non-linear regression (exponential) was used for curve fitting, the values of k_La could be read directly from Figure 5.4 for each aerator module. As indicated previously, the figure shows that the slope of the curves is much steeper for aerator module c compared to the other two for all liquid flow rates. In this case, the k_La values for module a, b, and c under 2.9 L/s, 3.1 L/s and 3.6 L/s effluent liquid flow rates were 0.123 min^{-1}(7.38 h^{-1}), 0.107 min^{-1}(6.42 h^{-1}), 0.143 min^{-1}(8.58 h^{-1}); 0.150 min^{-1}(9.00 h^{-1}), 0.127 min^{-1}(7.62 h^{-1}), and 0.159 min^{-1}(9.54 h^{-1}); 0.167 min^{-1}(10.02 h^{-1}), 0.150 min^{-1}(9.00 h^{-1}), and 0.216 min^{-1}(12.96 h^{-1}), respectively. It indicated that under all liquid flow rates, module c was the most efficient. However, with the same venturi injectors as with module c, module b obtained less oxygen through all experiments period even less than module a (with two venturi injectors).

Based on the value of k_La, a 1.25~1.44-fold increase in oxygenation capacity can be obtained by simply changing the way that three aerators are connected (module b and c), which is significant for practical use. It implies that friction can cause energy loss in the system and reduce the aeration efficiency. Therefore, it is wise to direct an increasing proportion of future research towards investigation of better designs of surface aeration piping system to reduce friction loss and achieve cost-effective aeration treatment of animal manure for odor control.

Table 5. 1 The performance of three modules under varied effluent liquid flow rate

Flow rate(L/s)	module	T (°C)	k_La(T) (h^{-1})	k_La(20) (h^{-1})	C_∞(T) (mg/L)	C_∞(20) (mg/L)	SOE (kgO_2/kWh)	SOTR (kgO_2/h)
	a	22.1	7.380	7.021	9.572	10.414	0.152	0.167
2.9	b	21	6.420	6.270	8.907	9.272	0.120	0.133
	c	22	8.580	8.183	9.416	10.204	0.173	0.190
	a	22	9.000	8.583	9.759	10.576	0.183	0.207
3.1	b	18.9	7.620	7.821	8.831	8.449	0.100	0.151
	c	17.8	9.540	10.051	9.827	8.996	0.137	0.206
	a	19.8	10.020	10.068	9.993	9.913	0.103	0.228
3.6	b	19	9.000	9.216	8.582	8.244	0.079	0.173
	c	20.5	12.960	12.807	10.248	10.456	0.139	0.305

Based on k_La, the values of aeration efficiency and oxygenation capacity for all the aerator modules were determined. A comparison among three modules under different effluent liquid flow rate, in terms of aeration efficiency, oxygenation capacity, and oxygen transfer coefficient is presented in table 5.1. High effluent flow rate may achieve better transfer rates, but at the expense of greater energy density and lower aeration efficiency (mass transfer per unit of power). With the liquid flow rate increased from 2.9, 3.1 to 3.6 L/s, the SOTR was 0.167, 0.133, 0.190 kgO_2/h for module a, 0.207,

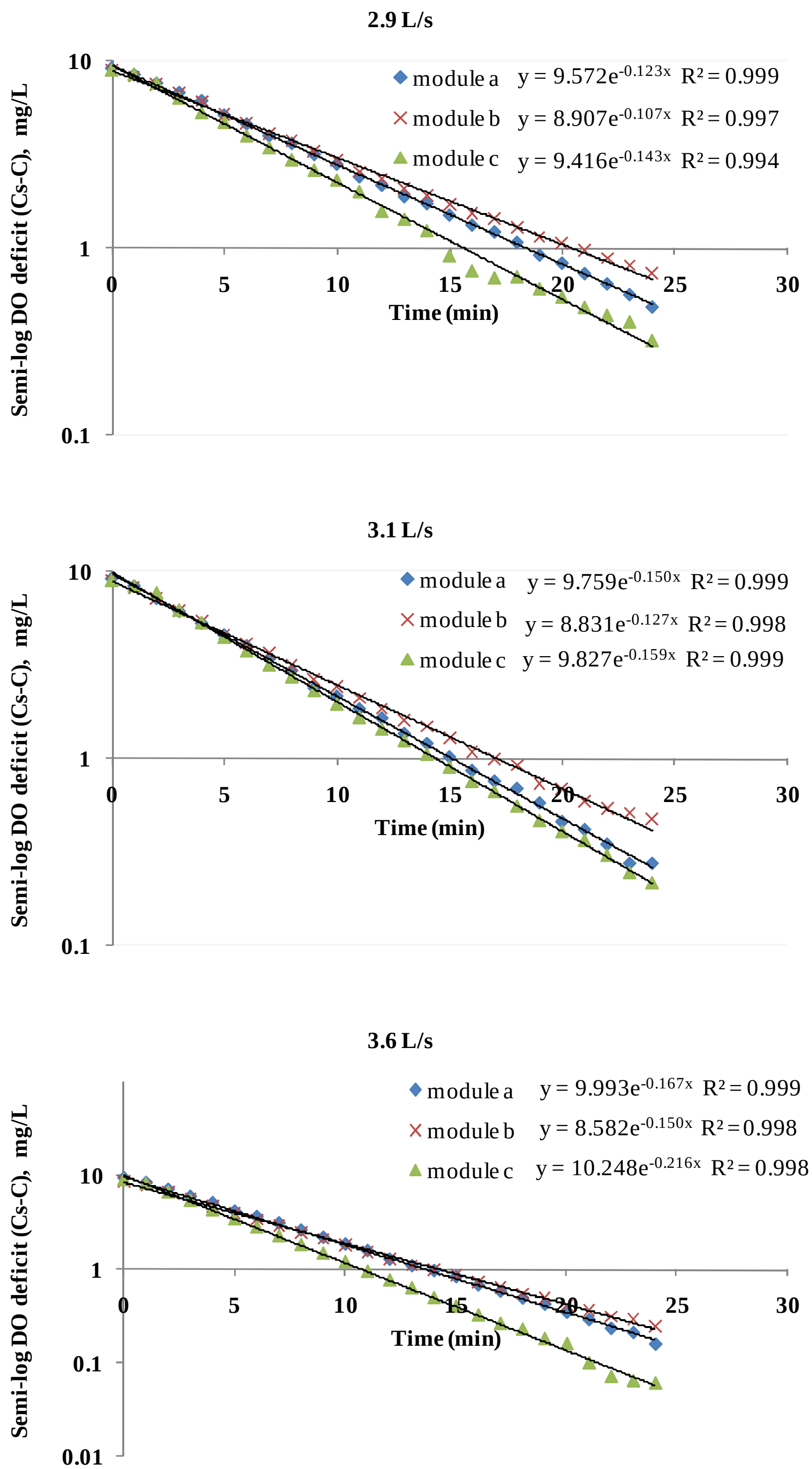


Figure 5. 4 Determination of oxygen transfer coefficients for the three aerator modules under different aerator module designs

0.151, 0.206 kgO_2/h for module b, and 0.228, 0.173, 0.305 kgO_2/h for module c, respectively. However, when the input power was considered and the liquid flow rate was increased from 2.9, 3.1 to 3.6 L/s, the aeration efficiency was 0.152, 0.120, 0.173 kgO_2/kWh for module a, 0.183, 0.100, 0.137 kgO_2/kWh for module b, and 0.103, 0.079, 0.139 kgO_2/kWh for module c, respectively.

5.3.2 The effect of liquid flow rate

5.3.2.1 Variation of dissolved oxygen with time for different liquid flow rates

In order to show the effect of liquid flow rate on oxygen transfer, a DO concentration profile was drawn as a function of time under varied liquid flow rates for three aerator module designs. Data from three runs of three aerator modules in the clean water test were averaged and presented in Figure 5.5, which depicted the changes in DO concentration during aeration. It shows a typical set of non steady-state reaeration curves, with the symbols representing the experimental data and the curves representing the model prediction. It is observed that the DO level in the aeration tank rises with time, such a rise in DO is very high at the beginning but subsequently it reaches a plateau, beyond which increase in DO level is very small, since the rate of change of DO concentration is a function of saturation deficit (Thakre, 2008). At start, the saturation deficit is high thus causing high rate change of DO concentration but with increase in aeration time, the saturation deficit decreases, resulting in decrease in the rate of change of DO concentration. It can also be noticed that an important increase of DO concentration with the increase of effluent liquid flow rate in the system. As clearly shown by the curves, the time required to reach saturation concentration decreases as

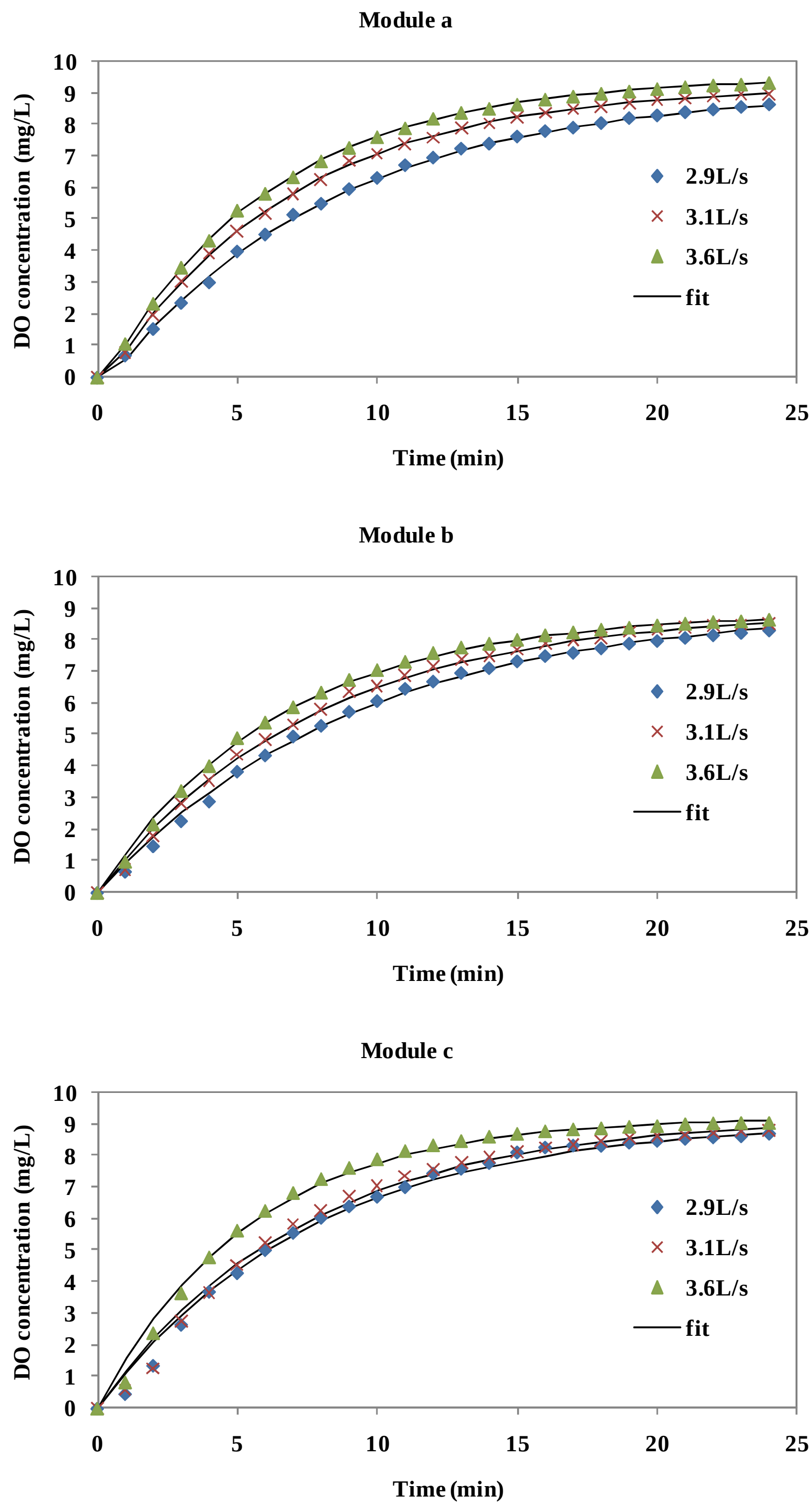


Figure 5. 5 Typical non steady state reaeration curves at varying effluent liquid flow rates

the liquid flow rate is increased. The aerator modules under 3.6L/s flow rate gave a better oxygen transfer capability than those under 2.9 L/s and 3.1 L/s flow rates in view of the fact that at any sampling point after aeration started, it transferred much more oxygen into the test liquid. The oxygen transfer efficiency appeared to deteriorate as the effluent liquid flow rate decreased in the system. Therefore it can be seen that as the effluent liquid flow rate increases, the time required to reach the maximum dissolved oxygen saturation concentration decreases. In other words with increase in effluent liquid flow rate, less time is required to attain maximum value of dissolved oxygen saturation concentration.

5.3.2.2 Effect of liquid flow rates on oxygen transfer

The oxygen transfer coefficient, k_La, represents the proportionality constant for the relationship between the rate of mass transfer and the concentration driving force. The determination of oxygen transfer coefficient is depicted as a function of the effluent liquid flow rate for three aerator modules in clean water in Figure 5.6, in which the slope of each linear regression line is the k_La for that particular aerator module. Since a non-linear regression (exponential) was used for curve fitting, the values of k_La could be read directly from Figure 5.6 for each aerator module. Therefore, the values at different flow rates were obtained after linear regression. All high R^2 values obtained (bigger than 0.99) indicated that dissolved oxygen concentration was well related to time after surface aeration operation. From Figure 5.6, it can be found that an increase in the effluent liquid flow rate leads to an increase in the oxygen transfer coefficient, practically so for all modules. When the effluent liquid flow rate was increased from 2.9, 3.1 to 3.6 L/s, the oxygen transfer coefficient in clean water test was 0.123, 0.150 and 0.167 min^{-1} for module a, and 0.107, 0.127 and 0.150 min^{-1} for module b, and 0.143, 0.159 and 0.216 min^{-1} for module c, respectively. On that account, it implies that the higher effluent liquid flow rate, the better oxygen transfer efficiency will be for all tested aerator module designs. Based on the value of k_La, a 1.36-, 1.40-, and 1.51-fold increase in oxygenation capacity can be obtained from 2.9 L/s to 3.6 L/s for module a, b and c, respectively. It implies that the effect of changing pump on module efficiency is obvious.

However, from Figure 5.7 showing the effect of liquid flow rate on the oxygen transfer coefficient under standard conditions, it was found that an increase in effluent liquid flow rate lead to an enhancement of the oxygen transfer coefficients, but the improvement was reduced if the effluent liquid flow rate went higher. Considering the trend of the liquid flow rate rising from 2.9 to 3.1 L/s and 3.1 to 3.6 L/s, it can be seen that in the range of relatively small liquid flow rates in this study (below 3.1 L/s), the influence of the effluent liquid flow rate on oxygen transfer coefficient is much stronger than that at large liquid flow rates for all test conditions. From Figure 5.7, under lower effluent liquid flow rate (from 2.9 L/s to 3.1 L/s), the oxygen transfer coefficient is enhanced by 11.12%, 12.38%, 11.42% per 0.1 unit for module a, b, and c, respectively and 11.64% on average. Under higher effluent liquid flow rate (from 2.9 L/s to 3.1 L/s), the oxygen transfer coefficient is averagely increased by 3.46%, 3.57%, 5.48% per 0.1 unit respectively and 4.17% on average.

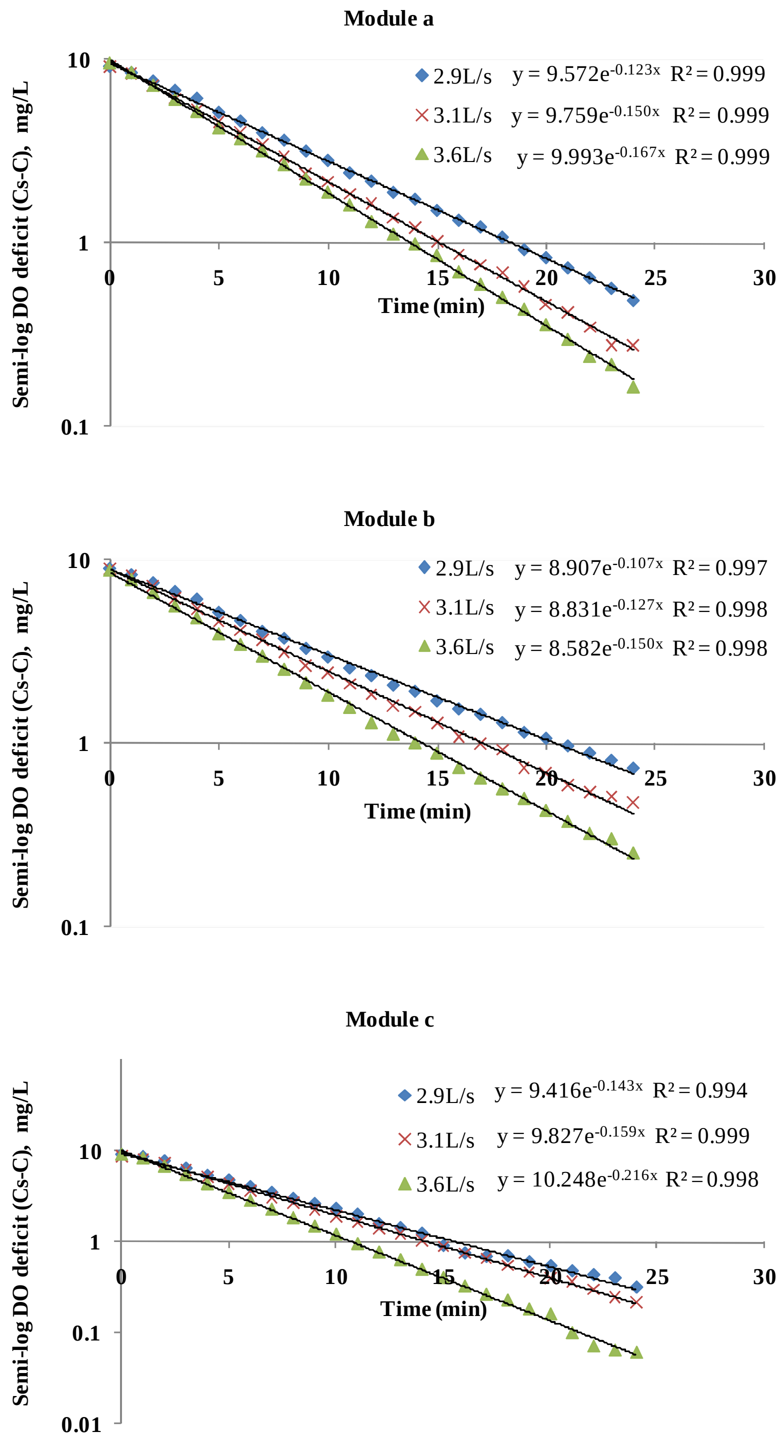


Figure 5. 6 Determination of oxygen transfer coefficients for the three aerator modules under different effluent liquid flow rates

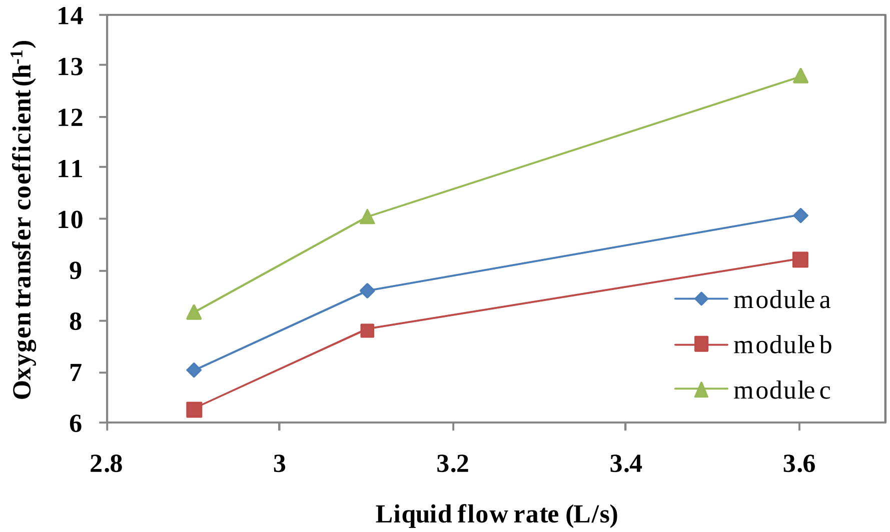


Figure 5. 7 Effect of liquid flow rate on the oxygen transfer coefficient under standard conditions

Venturi aeration is a high effective aeration method in water aeration systems that air is entrained into water from the injection orifice and the oxygen is dissolved into water, resulting from small air bubbles that are formed (Baylar et al. 2007). When air is introduced to water within the venturi injectors, it creates an air-water mixture or bubble flow. This occurs when a minimal amount of differential pressure exists between the inlet and outlet regions of the venturi device. According to the information from the injector vendor (Mazzei Injector Corp., Bakersfirld, Calif.), the particular model used in this study requires a liquid flow rate of 1.2 L/s to obtain the best air suction capability. In our design, the total maximal liquid flow rate that can be provided by the pump is 3.6L/s, indicating a sufficient liquid flow rate for each air injector, assuming even distribution of liquid among the three injectors. In the experiments, the venturi aerators tested for all runs meet the minimal differential pressure requirements. However, it is found that the oxygen transfer coefficient was not linearly related to the effluent liquid flow rate. The higher the effluent liquid flow rate, the lower improvement of oxygen transfer efficiency was observed. The residence time was taken into account to explain the results from Figure 5.7. Gillot et al. (2000) found that an increase of the bubble residence time in water, due to a neutralization of the liquid vertical convection movements by the horizontal flow, induced higher oxygen transfer efficiency. Therefore, in the venturi aeration system, when the mixture reaches the top, the lower effluent liquid flow rate has longer residence time, better mixing and less bubbles escaping into the atmosphere, which means more are entrained in the water in liquid phase. However, higher effluent liquid flow rate in the venturi aeration system causes higher mixture flow rate, more bubbles, and then a decrease in the oxygen transfer coefficient. On this account, it implies that there exists an optimal effluent liquid flow rate to make energy efficient in a venturi aeration system. Unfortunately, the results reported herein cannot give sufficient information on the optimal effluent liquid flow rate. Therefore, more work is warranted to further the effort in this area to determine the influence caused effluent liquid flow rate and residence time.

A semi-log regression model was used to analyze the data on oxygen transfer coefficient and flow rate in this study. Figure 5.8 showed the oxygen transfer coefficient as a function of the collected liquid flow rate for various configurations of the aeration system. The values of R^2 are 0.95, 0.94 and 0.99 for module a, b and c, respectively. The higher the R^2, the more closely the estimated regression fits the sample data. From the semi-log regression, it indicated that the lower the liquid flow rate, the greater its influence on oxygen transfer coefficient was. As could be expected, this response is likely due to a combined effect of decreased oxygen absorption during bubble formation and interference from adjacent rising bubbles. Besides, the increase in oxygen transfer should thus mainly be imputable to the neutralization of the vertical circulation of water (spiral flows) due to the increased velocity of each hole in the diffusion pipes (Gillot et al. 2000).

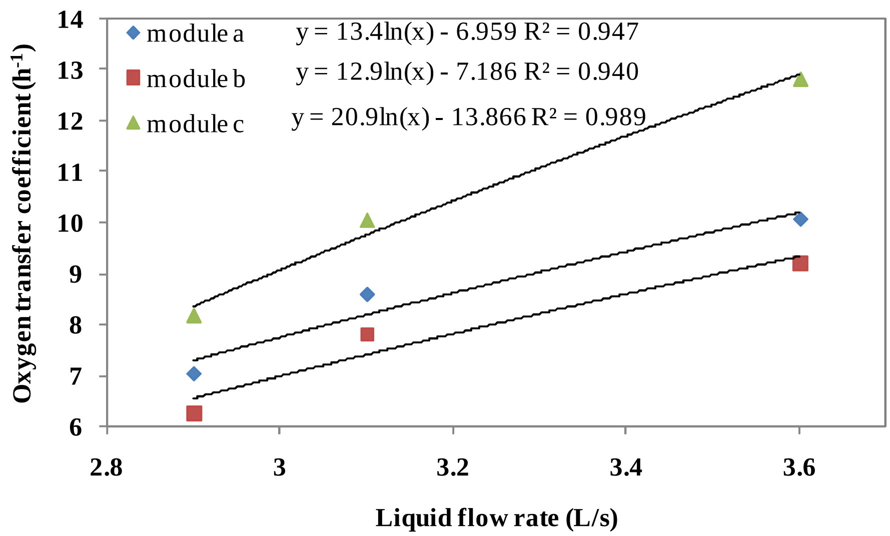


Figure 5. 8 Semi-log regression analyze between oxygen transfer coefficient and flow rate under standard conditions

5.3.3 Influence of liquid flow rate on alpha factor

The alpha factor indeed determines the degree of degradation of the substances responsible for oxygen transfer depletion, mainly surfactants. The alpha value ranges from approximately 0.2 to greater than 1.0 and is affected by many process water conditions including wastewater quality, intensity of mixing or turbulence, suspended solids concentration, method of aeration (ASCE, 1996). Another reason is the entrainment of foam created by surfactants. Such foams are either ephemeral or do not exist without surfactants. These foams can be stable and composed of fine bubble aggregates with high interfacial areas (Fan and Tsuchiya, 1990). Operations at these conditions have low aeration efficiency, since values of α approaching or higher than 1.0 may be reached at the sole expense of additional energy supplied to the system (Rosso et al., 2006).

Table 5. 2 Overall standard oxygen transfer efficiencies in clean water and under process conditions and corresponding alpha factors for different liquid flow rates

Liquid flow rate (L/s)	Aerator module	k_La (h^{-1})	k_La_p(h^{-1})	Alpha factor
2.9	Module b	6.270	4.446	0.71
	Module c	8.183	5.588	0.68
3.1	Module b	7.821	6.293	0.80
	Module c	10.051	7.755	0.77
3.6	Module b	9.216	8.074	0.88
	Module c	12.807	10.365	0.81

The overall standard oxygen transfer efficiencies in clean water and under process conditions and corresponding alpha factors for different liquid flow rate were presented in Table 5.2. The alpha value ranges from 0.71 to 0.88 for module b and 0.68 to 0.81 for module c under different liquid flow rate, respectively. The alpha factor is a decreasing function of the equivalent contact time between bubbles and the liquid phase. Capela et al. found that the higher the contact time (the lower liquid flow rate), the lower the oxygen transfer and the alpha value were, as the surfactants have more time to be absorbed to the interfacial area (Capela et al, 2002; Mueller et al. 2002; Rosso and Stenstrom 2006; Gillot et al, 2008). Similar results were obtained in this research for module b and c. This result can be explained by a higher contact time due to a lower upward velocity of the bubbles. Meanwhile, at a certain liquid flow rate, there is no big difference between two different modules at different effluent liquid flow rates. It implied that the alpha factor was not influenced by the aerator module design or configuration in this surface aeration system.

Moreover, it was found that the effect of foam created by surfactants on alpha factor was clear from the experiments. At a higher liquid flow rate, more foam was created in the tank, which caused lower aeration efficiency and higher alpha factor. The results and comments from Rosso et al. (2006) were presented here to explain the effect of foam on alpha factor. As time elapses, surfactant molecules migrate towards the surface, land, and hinder the interfacial renewal process, resulting in a steep and rapid decrease in value for the volumetric mass transfer coefficient $k_L a$. The volumetric mass transfer coefficient is the product of the velocity of reaction k_L and the interfacial area (a). Since bubbles are stable at smaller diameters in contaminated waters, the interfacial area is higher. Also, with increasing time, bubbles are losing mass by transferring oxygen to the liquid, hence their volume decreases and their specific area increases. Yet, the volumetric mass transfer coefficient $k_L a$ decreases, therefore k_L must decrease more rapidly than the interfacial area (a) increases. Eckenfelder and Barnhart (1961) observed that k_L decreases with increasing surfactant concentrations, but $k_L a$ is minimum at low surfactant concentrations, and partially recovers at higher concentrations. This is the effect of a larger interfacial area (a), but reduced velocity of reaction k_L.

5.3.4 Effect of surface aeration depth on oxygen transfer coefficient

Submerged aeration systems are used to increase DO levels and promote water circulation. Submerged diffusers release air or pure oxygen bubbles at depth and the ascending bubble plume entrains water, causing vertical circulation and lateral surface spreading. Oxygen is transferred to the water across the bubble interfaces as the bubbles rise from the diffuser to the water surface. Oxygen transfer also occurs across the air-water interface at the free surface due to turbulence induced by bubble-plume motion and water circulation. In a surface aeration system, the influence of the air-water transfer rate and the bubble-water transfer rates at various depths in the water column can be important in designing diffuser placement. The relationship between diffuser depth and oxygen transfer is also an important design criteria for a surface aeration system with distribution diffusers. By understanding where oxygen transfer occurs, diffuser systems can be designed to either maximize bubble transfer at certain water depths or to maximize surface turbulence and mixing which induces greater free surface transfer (DeMoyer et al., 2003). This part focuses on the effect of diffuser depths on oxygen transfer efficiency and oxygenation capacity.

Effect of diffuser depths on oxygen transfer coefficient is presented in Figure 5.8. The curve in Figure 5.8 indicates that, if the effluent liquid flow rate is kept constant, oxygen transfer coefficient is directly related to diffuser submergence (surface aeration depth). At 10cm submerges, the oxygen transfer coefficient was 0.357 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.357 h^{-1} to 0.275 h^{-1}, and then increased to about 0.413 h^{-1} at 40 cm depth. However, the oxygen transfer coefficient was decreased to 0.311 h^{-1} at 50 cm submerges. As shown in Figure 5.8, this phenomenon reflected the effect of surface aeration depth on oxygen transfer efficiency. More information on the effect of surface aeration depth on oxygen transfer efficiency will be presented in next chapter.

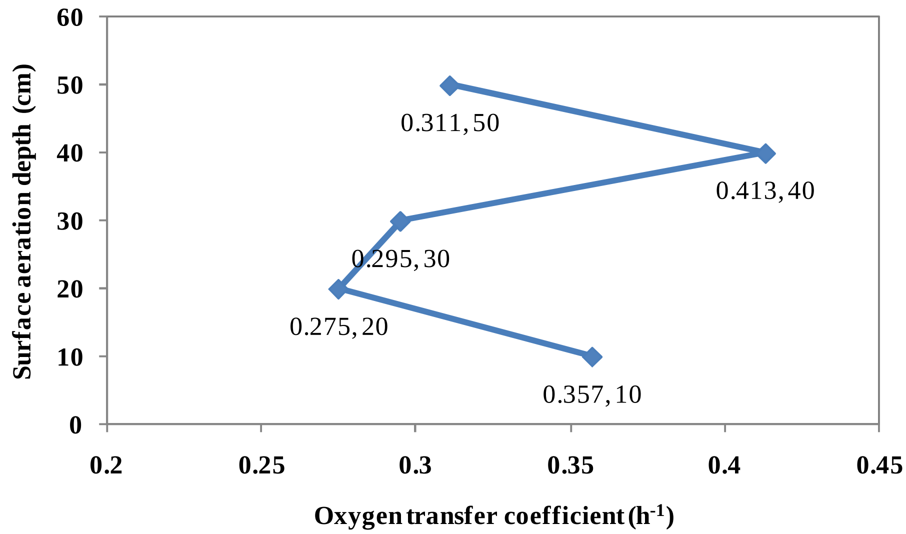


Figure 5. 9 Effect of surface aeration depth on the oxygen transfer coefficient of the system at constant effluent liquid flow rate

Moreover, the effect of increase in the surface aeration depth on aeration efficiency can be explained by the mechanism of exchange of the oxygen from gas to liquid. During this phase, the transfer is occurring by the processes of bubble formation, release, and ascension. This rate of oxygen transfer is dependent on the relative rate of ascent, bubble size, partial pressure of oxygen, temperature, and driving force (the difference between the liquid-film oxygen concentrations in equilibrium with the gas bubble and the bulk-liquid dissolved oxygen content) (Lioyed et al., 1979). Additionally, the oxygen transfer is influenced by the dispersion and coalescence characteristics produced through turbulent recirculation patterns existing in the tank (Michael et al., 1979).

With increasing the depth of water in the tank, one of these parameters such as the time is affected. The major effect is caused by the elongation of the path of bubbles in the tank. Since the ascension speed of bubbles is approximately constant, then a general increasing in depth of water will increase bubble residence time in the tank, which accordingly results in a longer time of bubble-water contact. Another effect of increasing the path of bubbles in the tank is specified by its ability to decrease the diameter of the bubbles, as the result of both absorption and breakup processes in bulk solution, which consequently will increase the oxygen transfer capacity of the system. Similar results also were reported by other researchers (Khudenko et al. 1986; Al-Ahmady, 2006). More discussions were presented in Chapter 6.

5.4 Conclusions

When surface aeration of swine manure lagoons is considered, the economical merit of power consumption is one of the most important factors for real applications. For this purpose, the effect of aerator design, effluent liquid flow rate, alpha factor and surface aeration depth on oxygen transfer efficiency in a surface aeration system with venturi injectors was studied with conclusions achieved below.

The k_La values for module a, b, and c under 2.9 L/s, 3.1 L/s and 3.6 L/s effluent liquid flow rate were 0.123 min^{-1}(7.38 h^{-1}), 0.107 min^{-1}(6.42 h^{-1}), 0.143 min^{-1}(8.58 h^{-1}); 0.150 min^{-1}(9.00 h^{-1}), 0.127 min^{-1}(7.62 h^{-1}), and 0.159 min^{-1}(9.54 h^{-1}); 0.167 min^{-1}(10.02 h^{-1}), 0.150 min^{-1}(9.00 h^{-1}), and 0.216 min^{-1}(12.96 h^{-1}), respectively. It indicated that under all liquid flow rates, module c was the most efficient. However, with the same venturi injectors as with module c, module b obtained less oxygen in all experiments, even less than module a (with two venturi injectors).

It implied that the higher the effluent liquid flow rate, the better the oxygen transfer efficiency were for all tested aerator module designs. Based on the value of k_La, a 1.36-, 1.40-, and 1.51-fold increase in oxygenation capacity can be obtained from 2.9 L/s to 3.6 L/s for module a, b and c, respectively. It implies that the effect of changing pump on module efficiency was obvious for better modules.

The alpha value ranges from 0.71 to 0.88 for module b and 0.68 to 0.81 for module c under different liquid flow rates, respectively. The alpha factor is a decreasing function of the equivalent contact time between bubbles and the liquid phase. Meanwhile, at a certain liquid flow rate, there is no big difference between two different modules at different effluent liquid flow rates. It implied that the alpha factor was not influenced by the aerator module design or configuration in this surface aeration system. Besides, at higher liquid flow rates, more foam was created in the tank, which in turn caused lower aeration efficiency and then higher alpha factor.

Oxygen transfer coefficient is directly related to diffuser submergence. At 10cm depth, the oxygen transfer coefficient was 0.357 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.357 h^{-1} to 0.275 h^{-1}, and then increased to about 0.413 h^{-1} at 40 cm. However, it decreased again to 0.311 h^{-1} at 50 cm depth.

Chapter 6 Effects of surface aeration depth on temperature profiles and aeration efficiency

6.1 Introduction

While most studies have focused upon lakes and reservoirs, where thermal stratification is the origin of the problem, fewer studies have focused upon an aerated lagoon, despite that submerged aeration systems are also used in swine manure lagoons to increase DO levels and promote water circulation, and thus reducing odor generation potential. Oxygen is transferred to the water across the bubble interfaces as the bubbles rise from the diffuser to the water surface. Oxygen transfer also occurs across the air-water interface at the free surface due to turbulence induced by bubble motion and water circulation. In a storage that is not well mixed, diffuser placement can influence the air-water transfer rate and the bubble-water transfer rate in the water column. The surface layer is assumed mixed thoroughly, thus uniform thermal and biochemical properties of water in this layer are presumed. However, the thickness of each layer changes that causes the layers to contract or expand respectively due to the surface aeration depth change. Moreover, the different surface aeration depth (diffuser submergence) also induces destratification in the lagoon.

The values and distribution of temperature in a mass of water play a fundamental role in the behavior of a surface aerated storage, in which the physicochemical and biological processes depend greatly on temperature. And the mixing phenomena are closely related to the thermocline, because thermocline restricts the heat and matter transport between the different layers defined in the water column as a consequence of thermal stratification. Aeration or circulations may disturb the temperature stratification of the lagoon. Therefore, the aim of this part is to understand the effect of surface aeration depth on temperature profile and aeration efficiency in a surface aerated tank with varied submergence diffusers. The experiments presented here are conducted to better understand some of the mixing processes involved and to provide data to obtain an optimal surface aeration depth. In order to better understand that aeration can increase oxygen content in the water and confirm that aeration is a useful tool to reduce odor generating potential, we also intend to research the influence of an artificial surface circulation layer on dissolved oxygen profiles and then compare the results from surface aeration with those from artificial surface circulation.

6.2 Materials and methods

A laboratory scale tank wrapped with foam and foil duct insulation (9.46 m^3) was used to carry out the experiments reported herein. The photos for the actual experimental system are presented in Figure 6.1. The schematic of the apparatus for determining aeration efficiency under different conditions is shown in Figure 6.2. The tank was equipped with a pump, a data acquisition system, dissolved oxygen probes, temperature probes, a heater and piping materials. The probes were calibrated immediately before and after each run. When operating, the centrifugal water pump circulated the liquid through the venturi air injection module to entrain air into the liquid.

For the study on the effect of surface aeration depth on aeration efficiency using a 9.46 m^3 volume tank, the aeration piping systems were placed at 100, 200, 300, 400 and 500 mm down from the water surface. DO concentrations were measured and recorded at top of the surface aerated layer for each placement of the piping systems. Compared with the smaller tank, a pump sized at 0.5 hp was used to achieve low flow velocity in the piping systems and avoid liquid turbulence in the tank.

Gu et al. (1995) found that the temperature difference between morning and evening could be up to 5°C at maximum in the surface layers and was less with depth and became very small (0 to 0.5°C) at the bottom in late summer/early fall in Minnesota ponds. In this part, the downward temperature mixing in the tank was taken into account. Therefore, a 4500 kW heater was used to warm up the surface water in the tank for two hours in the morning. The initial thermal stratification was established by electrically heating the surface water and gently circulating the warmed water to a lower depth. The bottom, top and the side wall of the tank were thermally insulated. A Campbell Scientific CR 1000 datalogger was installed and connected to a chain of 8 temperature probes. Absolute accuracy of the calibrated temperature probes was ±0.05°C (95% C.I.), and relative accuracy (the difference between successive measurements by the same probe) was ±0.01°C. Vertical temperature profiles were measured by 8 probes positioned at distances of 0.15, 0.5, 0.65, 0.8, 1, 1.25, 1.5 and 1.8 m. This probe arrangement gave information on the temperature stratification evolution in a vertical profile. The 8 temperatures were taken every 2 minutes, and every 10 minutes the preceding 8 measurements were averaged and stored. This measurement scheme was adopted to reduce the noise inherent to single measurements, while temporally resolving the fluctuations expected in the data. In this study, only the effect of different surface aeration depth on temperature profiles and aeration efficiency was taken into account.

The effect of wind and light on stratification was not considered although both of them could enhance or weaken the stratification in the field-scale. The detailed oxygen efficiency determination program was presented in Chapter 5.

Figure 6. 1 The pictures of the experiment tanks

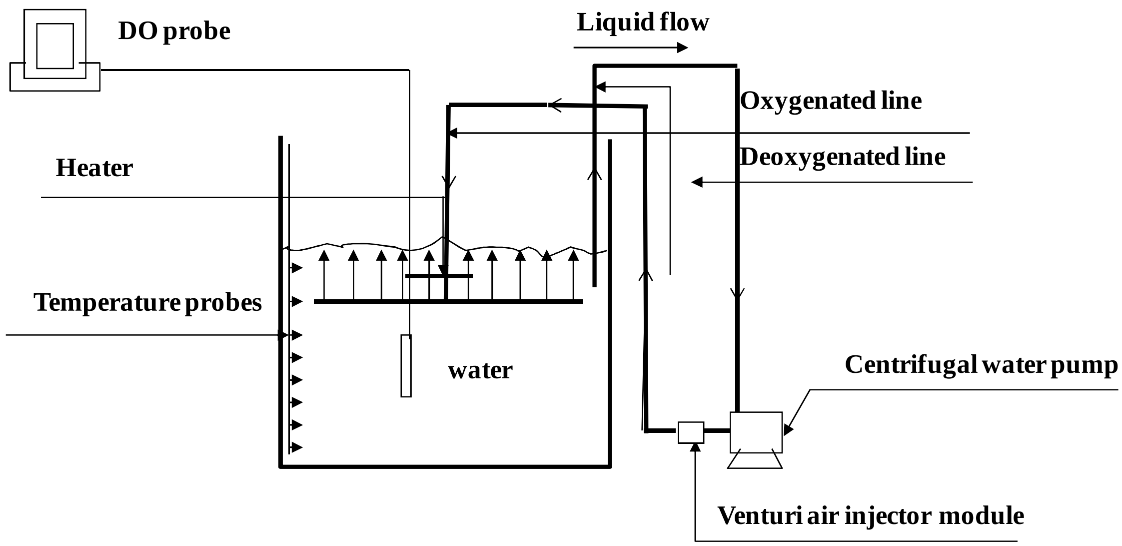


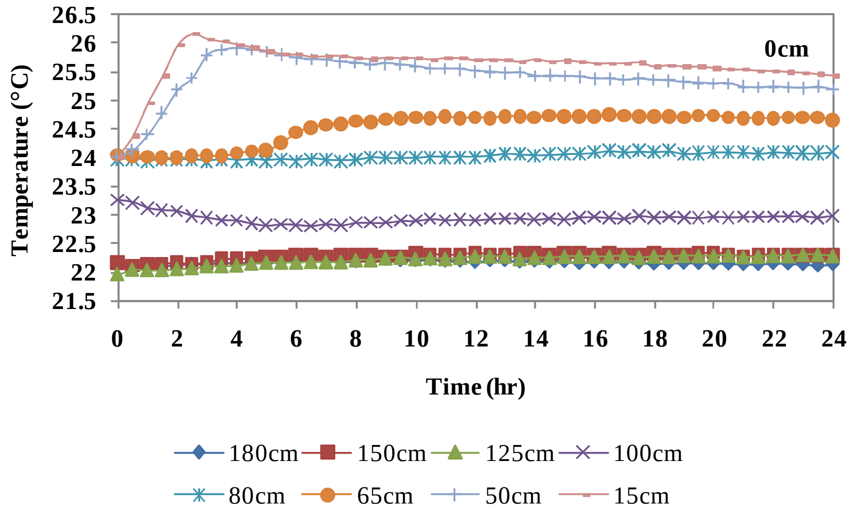
Figure 6. 2 Schematic of the apparatus for determining aeration efficiency of the aerators

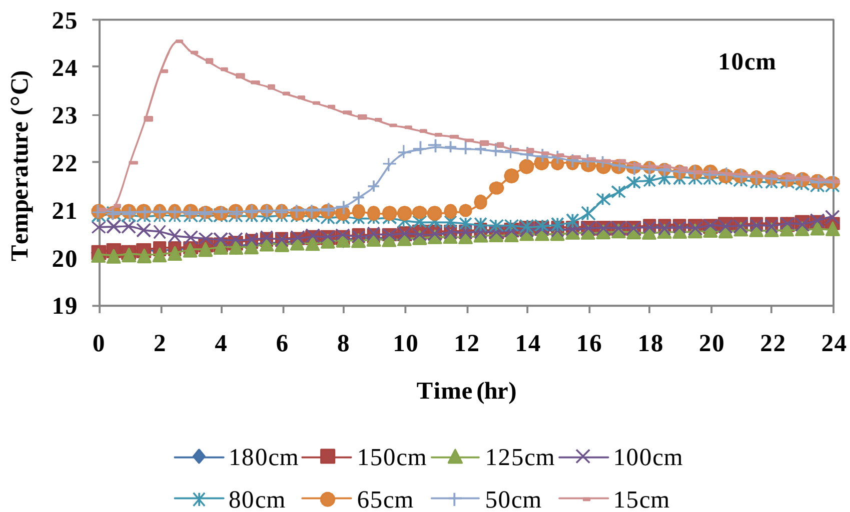
6.3 Results and discussions

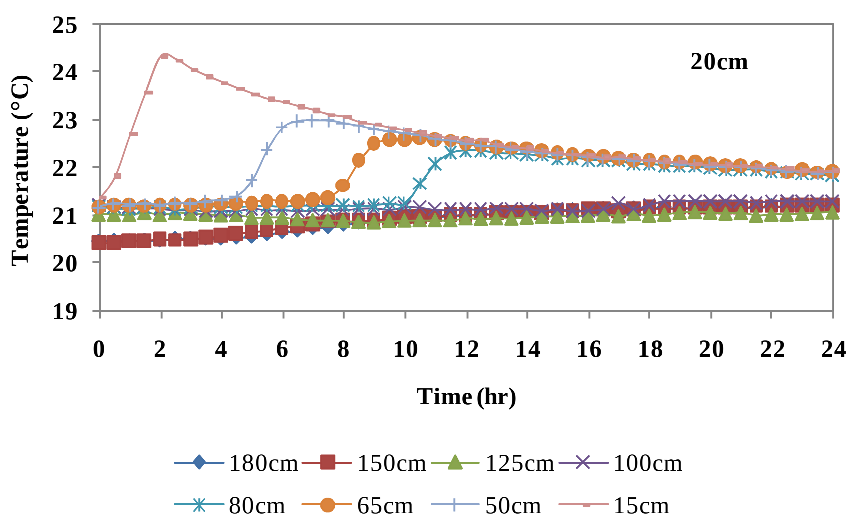
6.3.1 Temperature and stratification dynamics

Lagoon temperature profiles vary seasonally, with cool isothermal conditions in winter and warm thermally stratified daytime conditions in summer. In winter, the surface becomes cooler and denser than lower levels, leading to convective mixing, equalization of temperatures, and transport of DO from the upper levels so that adequate DO levels are maintained throughout the water body. But in summer, the surface becomes warmer than the lower levels so that convection stops, the water body becomes thermally stratified and DO becomes progressively depleted below the zone of wind and wave influence as a result of decomposition of organic matter. This process of thermal stratification can be prevented or reversed if circulation by mechanical means is provided during the summer period when natural convection does not occur. In autumn, when the upper levels of a thermally stratified water body become cooler and denser than the lower levels, a sudden “turnover” may occur, resulting in turbid, nutrient-rich water coming to the surface from the bottom, mixing through the whole water column. Turnover can be prevented by maintaining destratified conditions throughout the water column.

The water temperature profiles and time series measured in a surface aerated tank with varied diffuser submergences simulating summer conditions is shown in Figure 6.3. The highest line represented the surface temperature at 0.15 m depth, and the lowest line gave the water temperature near the tank bottom. Mixing, induced by surface aeration eventually led to partially uniform temperatures in the tank. The diurnal cycle of temperature dynamics, day-time stratification and night time mixing in the tank for all the tested conditions, was found from the experiments. When a liquid body is thermally stratified, two very different zones are found: the eplimnion, a layer closes to the surface, with a higher temperature and, therefore, with smaller density; and the hypolimnion, a layer closes to the bottom, with a lower temperature, thus denser (Dor et al., 1993). A mass flow between these two liquid layers is impaired by the density differences. As a result the volume occupied by the hypolimnion becomes a dead zone, decreasing the volume actually used for wastewater treatment. Thus, the effective volume of the tank in this study was defined as the volume of the surface completely mixed layer. A clear value for the thermal gradient that characterizes the occurrence of the stratification phenomenon is not found in the literature. Some investigations have used values that vary from 0.6 °C/m to 1.0 °C/m (Pires and Kellner, 1999). In this study a value of 0.6 °C/m was assumed to characterize the occurrence of thermal stratification.


0cm
Temperature (°C)
Time (hr)
180cm
150cm
125cm
100cm
80cm
65cm
50cm
15cm


10cm
Temperature (°C)
Time (hr)
180cm
150cm
125cm
100cm
80cm
65cm
50cm
15cm


20cm
Temperature (°C)
Time (hr)
180cm
150cm
125cm
100cm
80cm
65cm
50cm
15cm

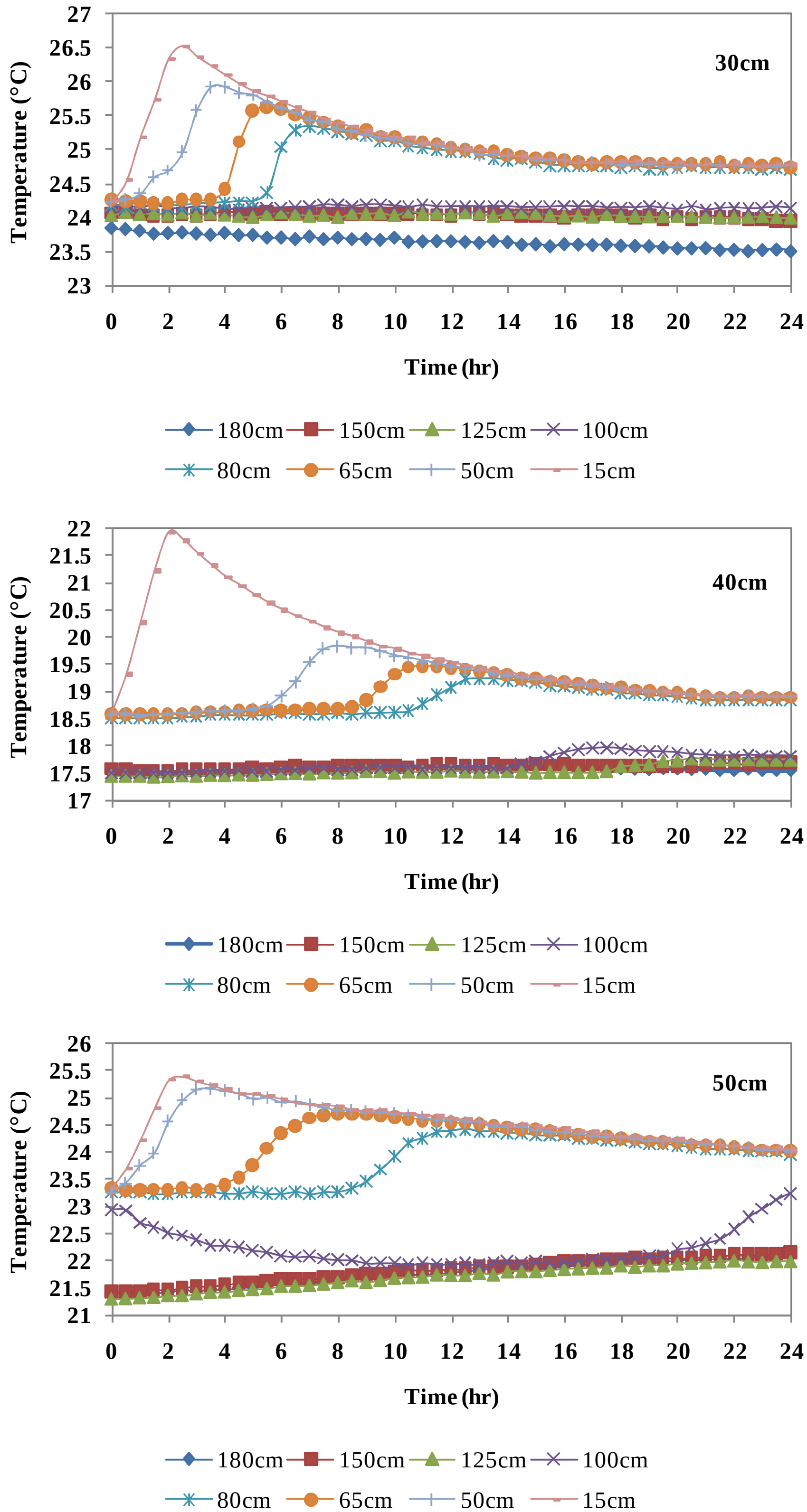


Figure 6. 3 Variations of water temperatures with depth and time

Several points may be identified: in the first stage, before turning on the heater and after night mixing, partial stratification was found. The temperature difference between top and bottom was around 1-3 °C. This is followed by the next stage (heater was on) where the temperature between the upper and lower layer was increased. The surface layer (10-30 cm) of the tank was heated and caused stratification between the top and lower layers and more stratified upper layers were found in this stage. It can be seen from the Figure 6.3 that the temperature in the upper layers increased gradually. Third, the density difference is due to the difference in temperature in the tank. During the night (after heater was off in this study), due to surface cooling and convective natural mixing, the tank temperatures were fairly uniform for the top 80cm layer. As the temperature profiles for depths at 10, 20 and 40 cm were almost the same at initial surface water temperature of 18-21 °C, this mixing was not diffuser induced. Fourth, heat can penetrate deeper into the tank when the surface temperature was more than 24 °C (no aeration or 30 cm surface aeration) or surface aeration depth was 50 cm, thus resulting in weaker stratification in the top layers compared with the initial surface water temperature that was lower than 24 °C. The bigger order of magnitude of the vertical heat transfer was the most likely cause. Last, the temperature profile without aeration (0 cm labeled in Figures) was a little different from the other ones. The upper layers were not well mixed after cooling period and strong stratification was found. Therefore, destratification was observed in the top layer by low flow rate surface aeration.

The surface temperature, bottom temperature and temperature difference at different surface aeration depths are presented in Figure 6.4 and Table 6.1. It gave the initial temperature difference for the different surface aeration depth run.

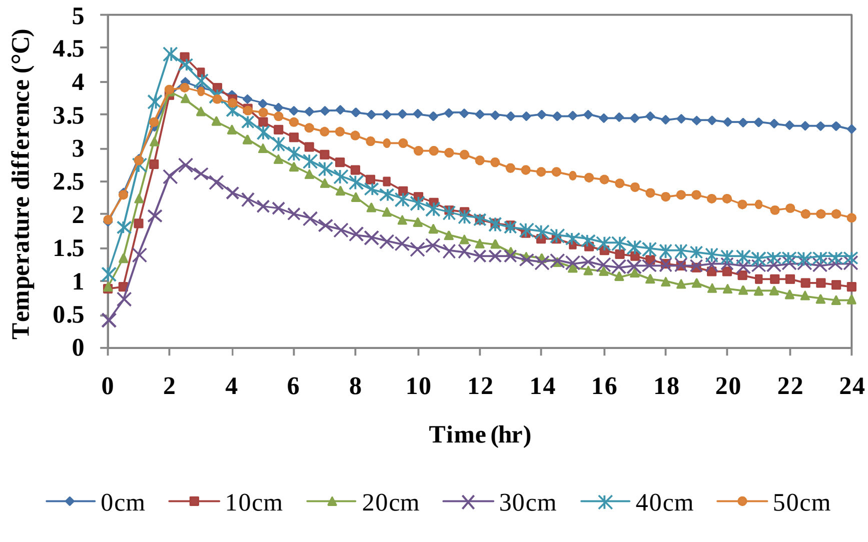
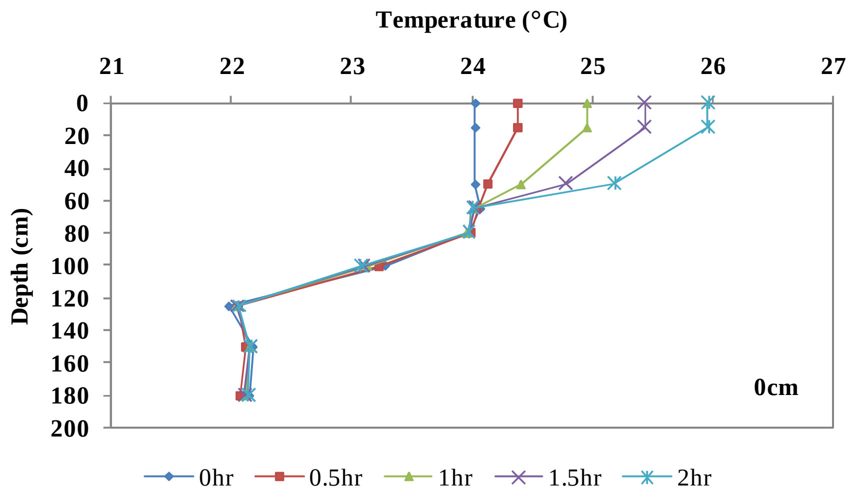
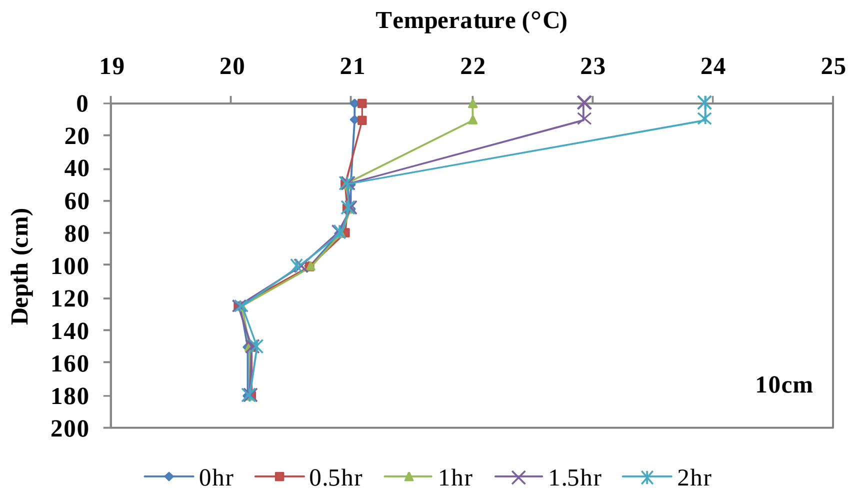


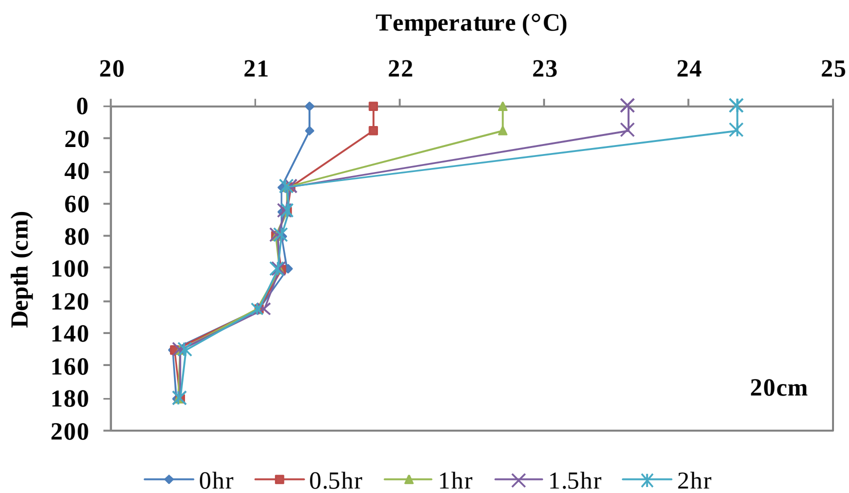
Figure 6. 4 The temperature difference vs time at varied surface aeration depths

Table 6. 1The time series temperature conditions

		0hr	2hr	4hr	8hr	12hr	16hr	20hr	24hr
	T surface	24.02	25.96	25.97	25.73	25.69	25.64	25.56	25.43
	T differece	1.88	3.82	3.81	3.54	3.51	3.46	3.40	3.29
0cm	T bottom	22.14	22.14	22.16	22.19	22.18	22.19	22.16	22.14
	T surface	21.02	23.94	23.97	23.06	22.48	22.07	21.81	21.61
	T differece	0.89	3.79	3.73	2.68	1.93	1.47	1.15	0.92
10cm	T bottom	20.13	20.15	20.24	20.38	20.55	20.60	20.66	20.69
	T surface	21.37	24.33	23.79	23.07	22.57	22.26	22.07	21.94
	T difference	0.92	3.86	3.29	2.27	1.57	1.16	0.89	0.73
20cm	T bottom	20.45	20.47	20.51	20.80	21.00	21.10	21.18	21.21
	T surface	24.24	26.35	26.10	25.40	25.03	24.85	24.81	24.78
	T differece	0.40	2.57	2.32	1.69	1.37	1.23	1.26	1.26
30cm	T bottom	23.85	23.77	23.78	23.71	23.66	23.62	23.56	23.52
	T surface	18.62	21.93	21.12	20.08	19.54	19.19	18.97	18.92
	T differece	1.10	4.43	3.58	2.48	1.93	1.57	1.38	1.35
40cm	T bottom	17.52	17.51	17.54	17.60	17.61	17.62	17.60	17.57
	T surface	23.35	25.33	25.16	24.85	24.62	24.44	24.25	24.05
	T differece	1.92	3.89	3.67	3.18	2.82	2.53	2.23	1.95
50cm	T bottom	21.43	21.44	21.48	21.67	21.80	21.91	22.02	22.09


Temperature (°C)
21 22 23 24 25 26 27
Depth (cm)
0 20 40 60 80 100 120 140 160 180 200
0cm
0hr 0.5hr 1hr 1.5hr 2hr


Temperature (°C)
19 20 21 22 23 24 25
Depth (cm)
0 20 40 60 80 100 120 140 160 180 200
10cm
0hr 0.5hr 1hr 1.5hr 2hr


Temperature (°C)
20 21 22 23 24 25
Depth (cm)
0 20 40 60 80 100 120 140 160 180 200
20cm
0hr 0.5hr 1hr 1.5hr 2hr

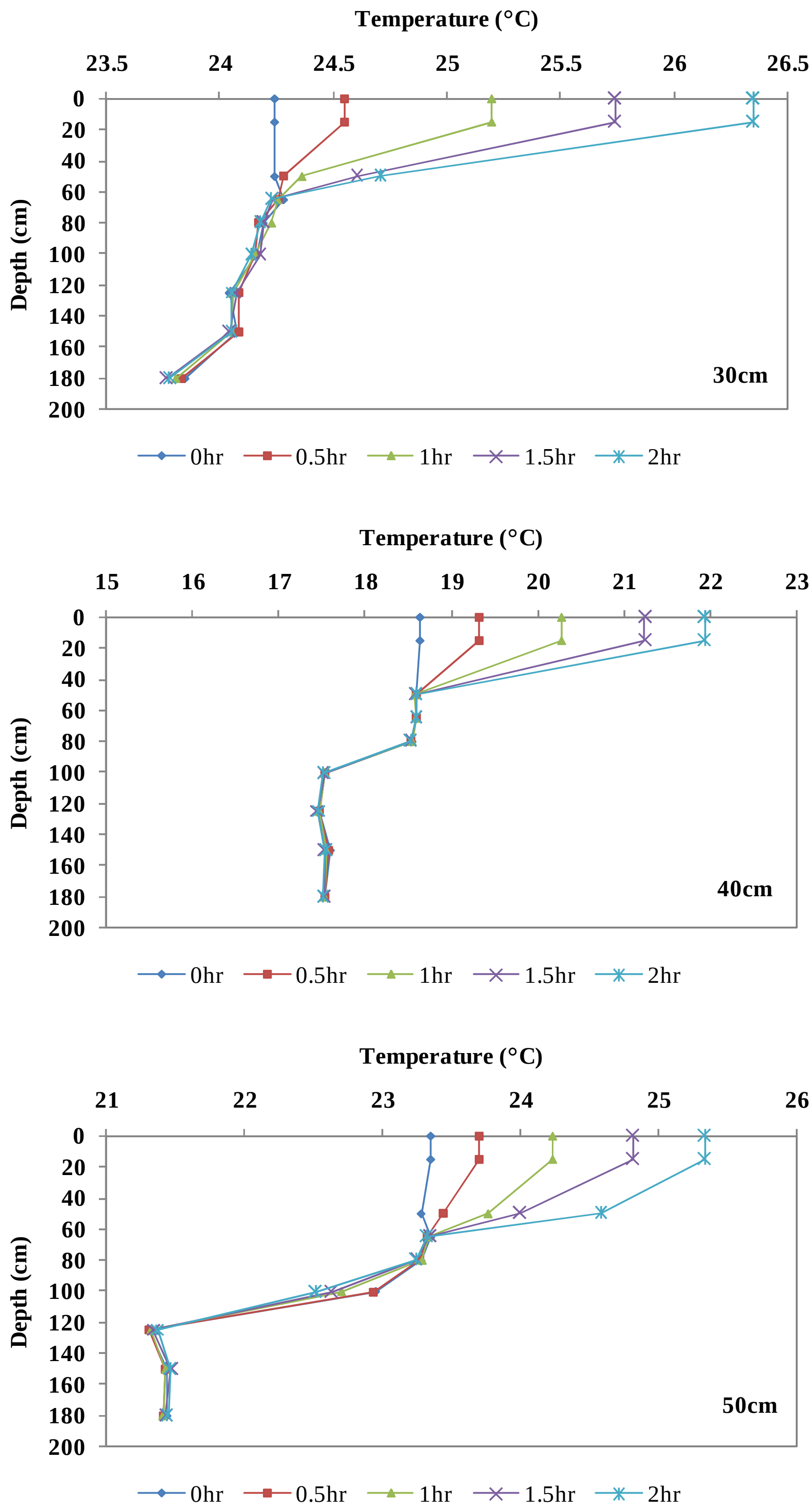
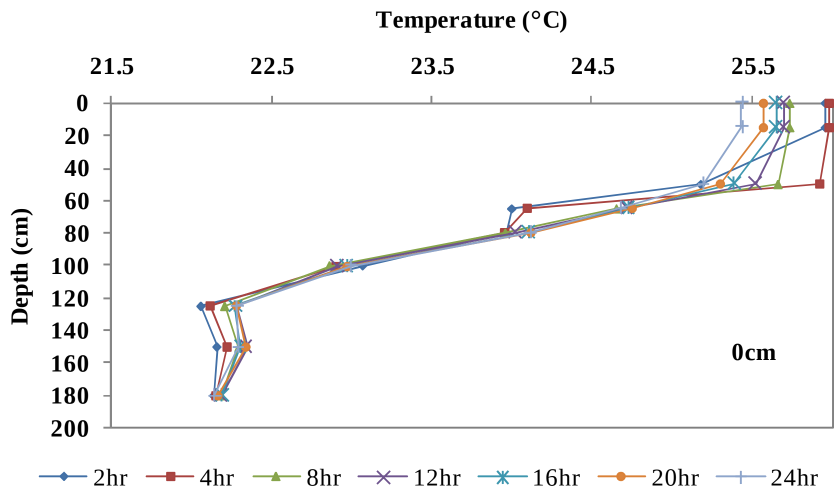


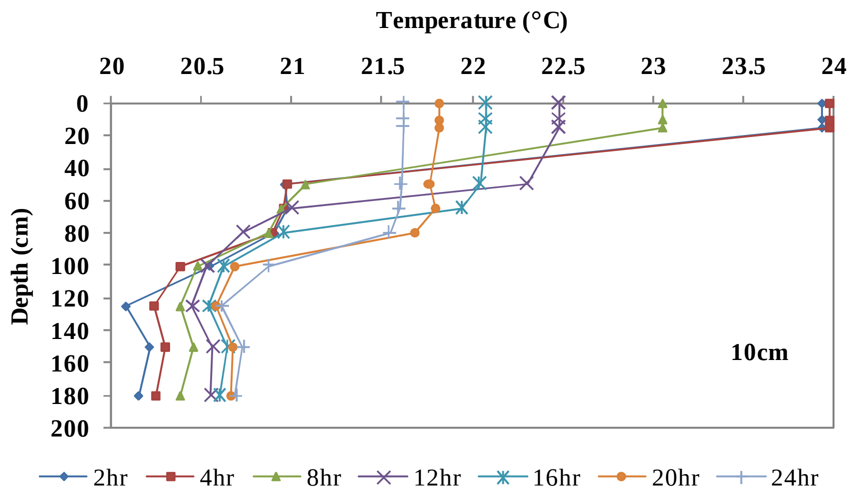
Figure 6. 5 Vertical temperature profiles with depth and time at heating

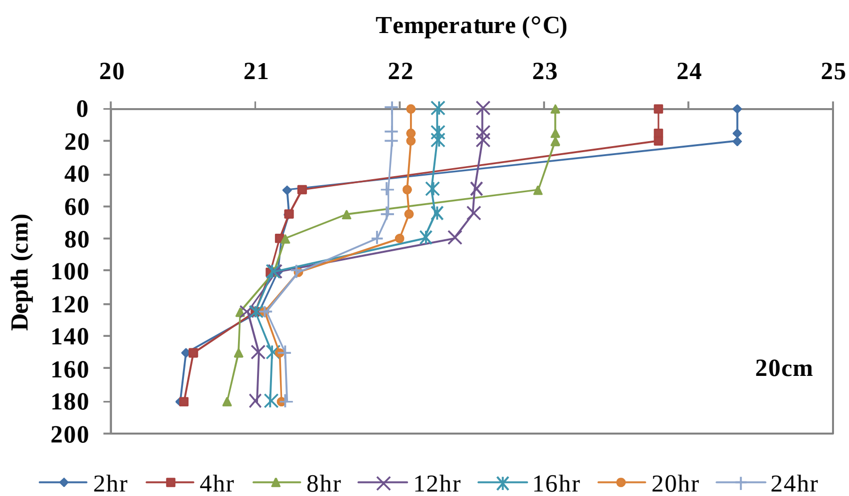
Measured mixed layer depths were plotted over time at heating period in Figure 6.5 and at cooling period in Figure 6.6. Mixed layer depth values were determined from temperature profiles based on a temperature decrease of 0.6 °C from the value nearing the surface (T(zm)=T(0)- 0.6 °C). This value was chosen as an intermediate value, greater than the measurement noise, but much less than the change in temperature across the thermocline (Herb et al., 2005). Figure 6.5 gave measured temperature profiles for all surface aeration depths at 30 minutes time increments at heating period. The temperature above the diffuser was observed to be uniform due to a mixed layer, which was formed because of surface aeration. And there was a strong temperature gradient between the submergence and 50 cm or 65 cm. No effect on temperature profile was found below 65 cm water depth from top.

Several points can be made from Figure 6.5 and 6.6. First, the variation of surface heat fluxes produced a significant daily variation in the mixed layer depth. During surface heating (positive heat input), the mixed layer depth was reduced to a minimum value that depended on heat flux. At night, heat loss through a surface increased the mixed layer depth rather uniformly, independent of surface aeration depth. Second, as the effluent piping was closer to the water surface than the influent piping in the same tank, the inflow added warmer water to the surface aerated layer. This can enhance stratification in summer. Third, the inflow caused some turbulent mixing due to the flow patterns that it set up (diffuser entrainment, plunging flows, spreading flows, etc.). However, this mixing caused by small flow-rate surface aeration diffuser systems was not significant compared with the influence of temperature. For 10, 20, 30, 40 50 cm surface aeration depth, the deeper diffuser submergence didn’t induce weaker stratification. The mixed layer, a strong gradient temperature layer and bottom layer were found for all surface aeration depths, which implied that this mixing was not diffuser system induced. The weather may have a much stronger influence on the stratification dynamics of the tank than diffuser mixing (forced convection). Last, as bubbles were efficient at transporting the lower layer liquid into the upper layer and thereby increasing the uniform density at the top layer in the tank, the mixing time for the upper 50 cm layer was different. The 50 cm surface aeration depth needed the shortest time to obtain a uniform mixed 50 cm layer, while the 10 cm surface aeration depth took the longest. It implied that the surface aeration system could enhance the mixing above the diffuser but surface cooling and convective natural mixing affecting the tank temperature might be more important.

Similar results were reported by other researchers. Chen et al. (2000) did research on the mixing of liquids by a plume of low-reynolds number bubbles. They found that very little dense liquid from the lower layer is transported across the interface by the rising bubbles. The density in the upper layer therefore remained constant and equal to that at the beginning of the mixing process. Lawson et al. (2007) found that the uppermost 2 m of the water column experienced marked diurnal heating and cooling, while below 2 m or so the temperature was more influenced by longer-term weather patterns, seasonal changes, and mixing. Gu et al. (1996) studied on inflow and jet mixing effects on water quality stratification in shallow wastewater stabilization ponds and the results showed that wastewater stabilization ponds, although only 1 to 2 m deep, stratify and destratify intermittently depending primarily on weather. Other conclusions made by Gu et al. (1996) were presented as follows. Stratification developed primarily by differential


Temperature (°C)
21.5
22.5
23.5
24.5
25.5
0
20
40
60
80
100
120
140
160
180
200
Depth (cm)
0cm
2hr
4hr
8hr
12hr
16hr
20hr
24hr


Temperature (°C)
20
20.5
21
21.5
22
22.5
23
23.5
24
0
20
40
60
80
100
120
140
160
180
200
Depth (cm)
10cm
2hr
4hr
8hr
12hr
16hr
20hr
24hr


Temperature (°C)
20
21
22
23
24
25
0
20
40
60
80
100
120
140
160
180
200
Depth (cm)
20cm
2hr
4hr
8hr
12hr
16hr
20hr
24hr

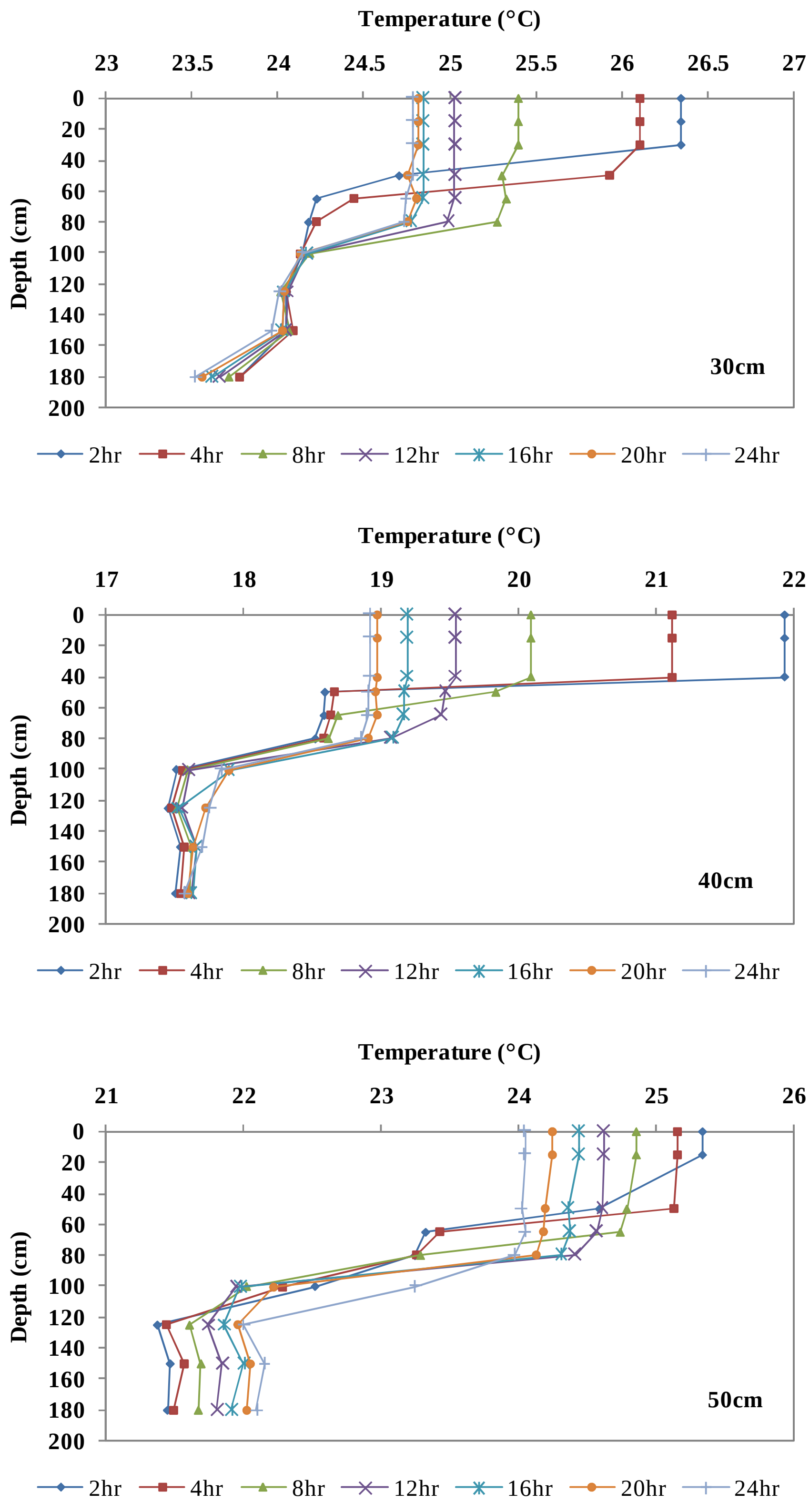


Figure 6. 6 Vertical temperature profiles with depth and time at cooling

heating of the pond water through its surface and, in the absence of artificial aeration or mixing devices, by insufficient wind mixing. Inflows from pipes produced some jet mixing effects, but in the pond system studied these effects were not sufficient to destroy stratification. Weather effects dominated stratification dynamics. Solar radiation caused day-time heating of the surface layers and was the main reason for temperature stratification of the ponds. A completely or partially mixed state was often brought about by night-time cooling. Naturally, more mixing occurred during period of stronger winds. Temperature stratification was also affected by seasonal variations in weather. Three types of temperature stratification behavior occurred from April to October: continuously well-mixed conditions, stratification during day and complete mixing during night, and continuous stratification over several consecutive days and nights. Stratification occurred more than 50 percent of the time.

6.3.2 Effect of diffuser submergence on vertical diffusion coefficient

Density stratification due to vertical temperature gradients inhibits vertical mixing and the mixing in turn affects the distribution of nutrients, and other water quality constituents. Direct measurements of vertical turbulent diffusion are not easy because of the three-dimensional nature of the diffusion field, and the spatial and temporal scales. To estimate diffusion values, one can rely on measurements of water temperatures, which is the most commonly used method because of its simplicity (Hondzo et al., 1993). The purpose of this part was to estimate vertical eddy diffusion based on water temperature measurements for different surface aeration depths.

In the model the tank is described by a group of horizontal layers, each of which is well mixed. Vertical transport of heat is described by a diffusion equation in which the vertical diffusion coefficient $K_z(z)$ (m^2/day) is incorporated in a conservation equation of the form (Henderson-Sellers, 1984; Chapra, 1997):

$$A\frac{\partial T}{\partial t} = \frac{\partial}{\partial z}(K_z A\frac{\partial T}{\partial z}) + \frac{H}{\rho c_p} \qquad (1)$$

Considering a non heat flux boundary condition after cooling, vertical diffusion coefficient K_z can be estimated by:

$$K_z = \left(\frac{\partial T}{\partial z}\right)^{-1} \int_h^z \frac{\partial T}{\partial t} dz \qquad (2)$$

Where T(z,t) is water temperature in °C as a function of depth (z) and time (t), A(z) is the horizontal area of the storage (m^2), H is the heat source or sink (kcal m^{-2} day^{-1}) , ρ is the water density (kg/m^3), and c_p is the specific heat of water (cal g^{-1} $°C^{-1}$).

Flow mechanisms contributing to vertical transport and mixing are currents, breaking internal waves and internal shear instability or natural convection due to cooling. In the model vertical transport is simulated by turbulent vertical diffusion so the vertical diffusion coefficient K_z (Equation 3) has to be specified. The parameters to which K_z can be related are lagoon surface area, A, maximum depth, h, strength of stratification represented by Brunt-Vaisala frequency, N, and wind speed, W.

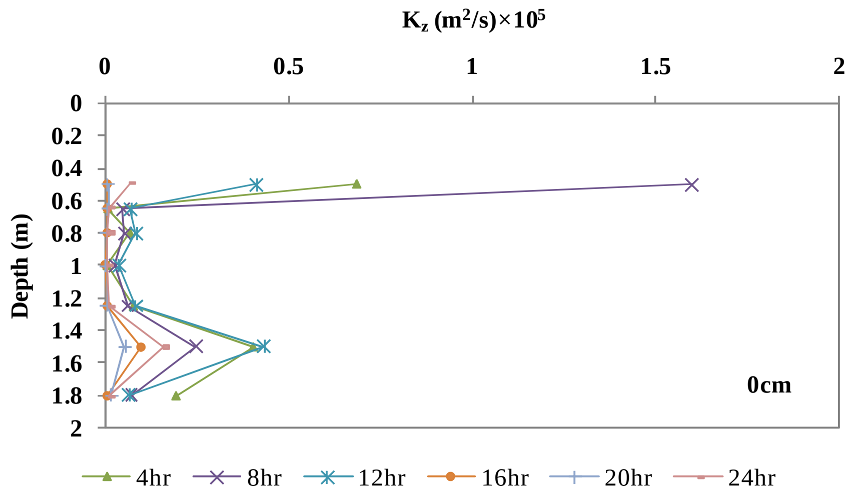
Boundary conditions are specified at the water surface (z=0) and the sediment water interface (z=h).

At z=h $$\frac{\partial T}{\partial z} = 0 \tag{3}$$

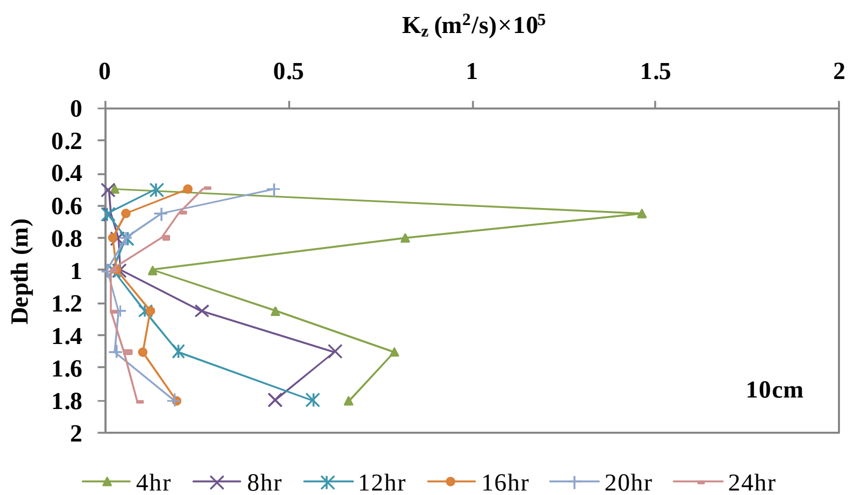
At z=0 $$K_z \frac{\partial T}{\partial t} = H \tag{4}$$

When a horizontal layer of water is subjected to surface cooling, a cool thin layer formed at the surface breaks up periodically, generating sinking sheets of cool water which force a return flow toward the surface. The thermal movements are active as long as cooling continues and provide a vertical stirring mechanism that keeps the horizontal layer nearly well-mixed vertically except for the very thin surface cool boundary layer.

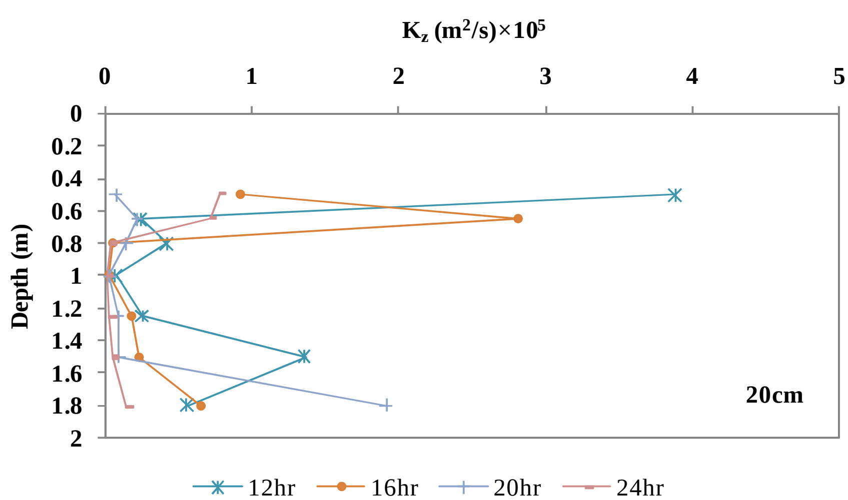
Vertical diffusion coefficients calculated are presented in Figure 6.7 and 6.8. Vertical diffusion coefficient values at upper layers (above 0.8m) ranged from approximately 0.002 to 0.160 cm^2/s, 0.004 to 0.146 cm^2/s, 0.003 to 0.388 cm^2/s, 0.009 to 0.692 cm^2/s, 0.002 to 0.510 cm^2/s, and 0.001 to 0.533 cm^2/s with an average 0.017, 0.023, 0.066, 0.120, 0.083 and 0.148 cm^2/s at surface aeration depth 0, 10, 20, 30, 40 and 50 cm, respectively. The highest vertical diffusion was found at 50cm surface aeration depth, while zero aeration had the lowest values. Besides, all the experiments showed that the vertical diffusion coefficient values were infinite above 50 cm depth and it indicated that a well mixed 50 cm layer was observed for all surface aeration depths. Between 50 cm and 80 cm depth, there was a strong gradient temperature profile. The vertical diffusion coefficient at 1 m was around zero for most conditions and very weak vertical diffusion coefficient was between 0.8 m to 1.25 m. It implied that no vertical diffusion observed at 1 m depth. Below 1.25 m, there was a stronger diffusion than the layer



Kz (m²/s)×10⁵
0
0.5
1
1.5
2
Depth (m)
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
1.8
2
0cm
4hr
8hr
12hr
16hr
20hr
24hr



Kz (m²/s)×10⁵
0
0.5
1
1.5
2
Depth (m)
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
1.8
2
10cm
4hr
8hr
12hr
16hr
20hr
24hr



Kz (m²/s)×10⁵
0
1
2
3
4
5
Depth (m)
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
1.8
2
20cm
12hr
16hr
20hr
24hr

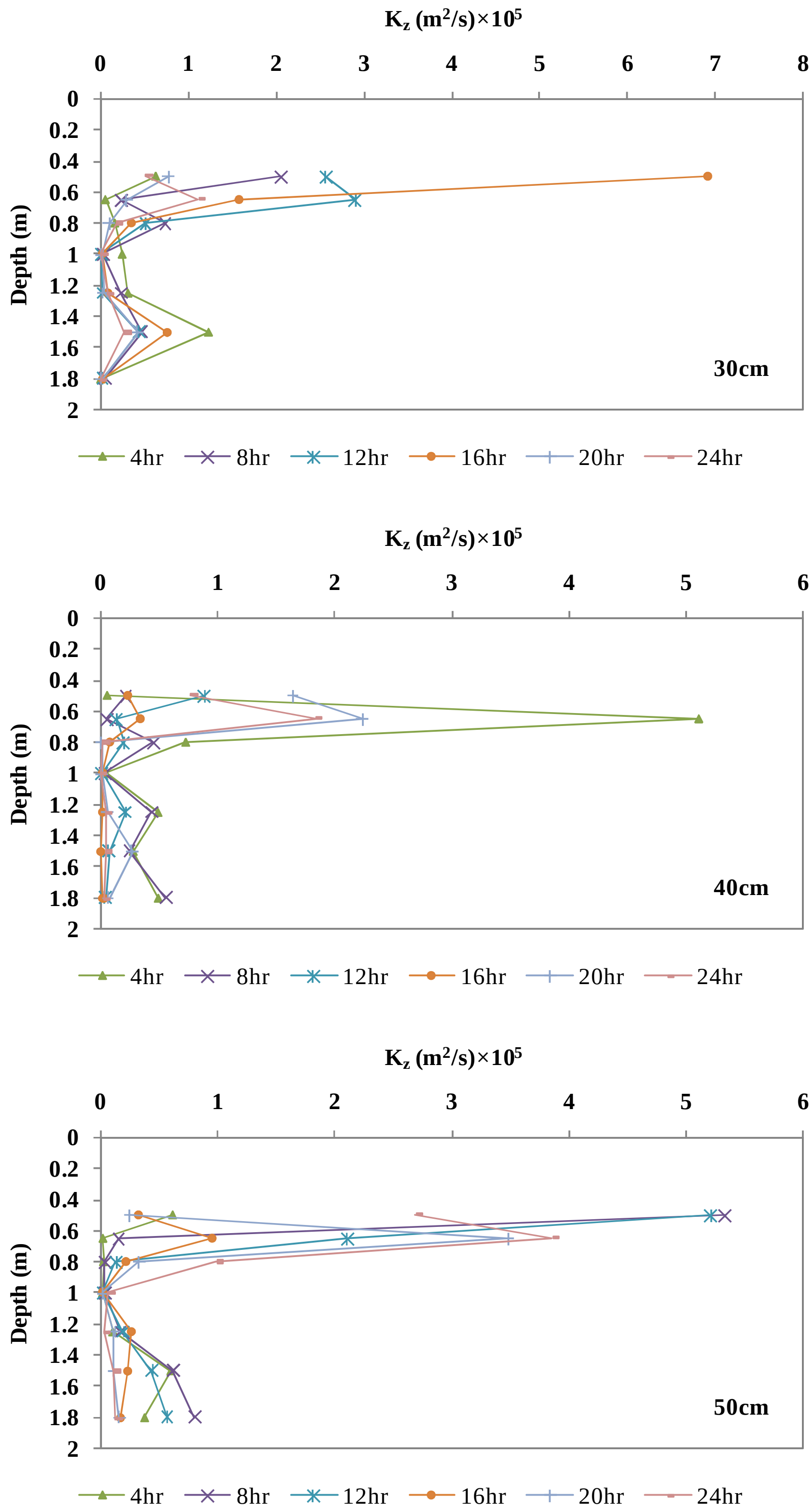
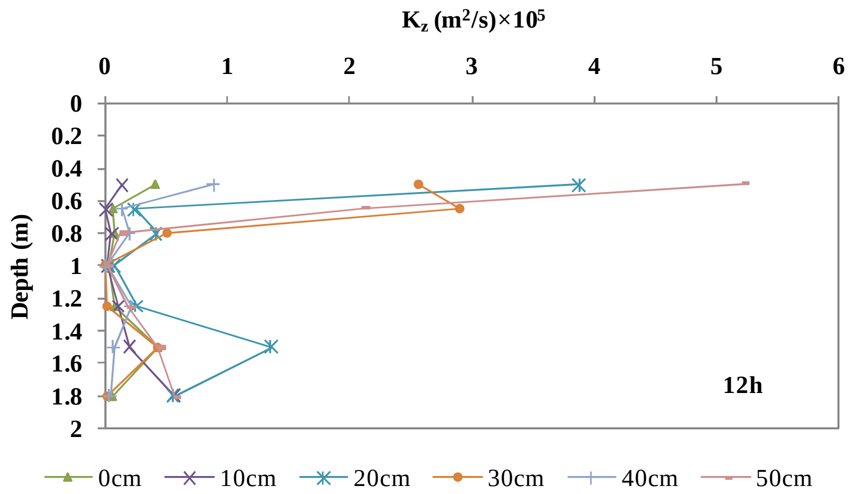
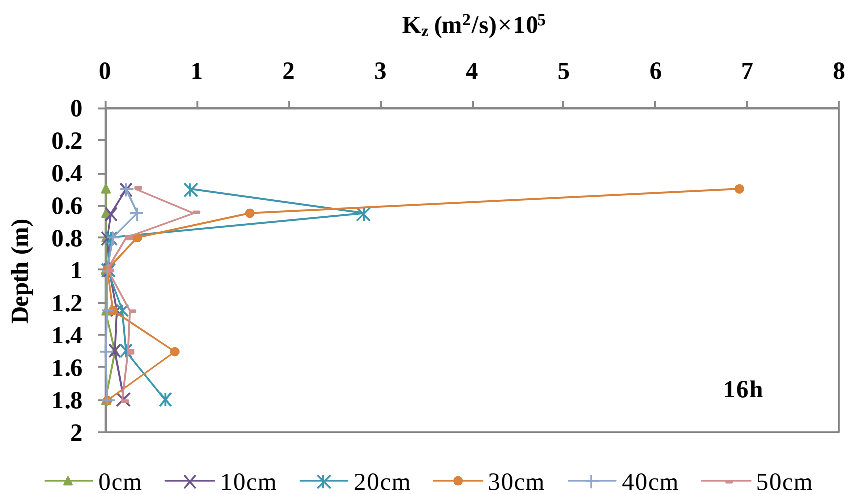


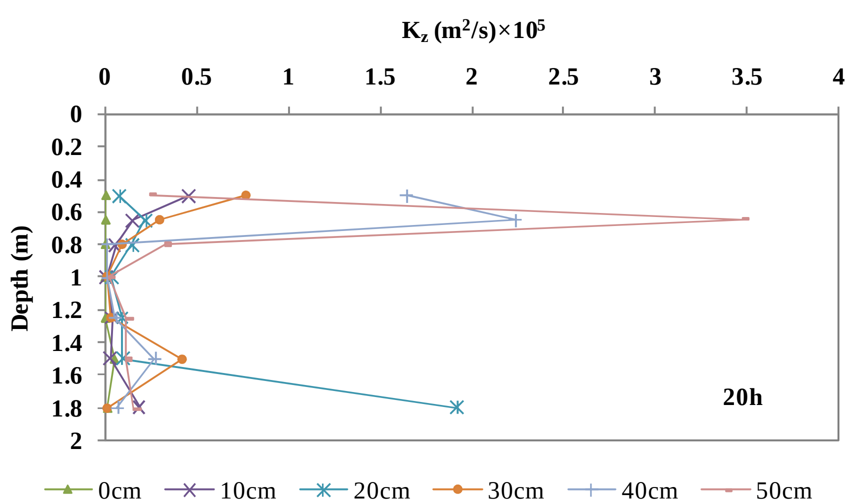
Figure 6. 7 Vertical diffusion coefficient time series at different diffuser depths



K_z (m²/s)×10⁵
Depth (m)
12h
0cm
10cm
20cm
30cm
40cm
50cm



K_z (m²/s)×10⁵
Depth (m)
16h
0cm
10cm
20cm
30cm
40cm
50cm



K_z (m²/s)×10⁵
Depth (m)
20h
0cm
10cm
20cm
30cm
40cm
50cm

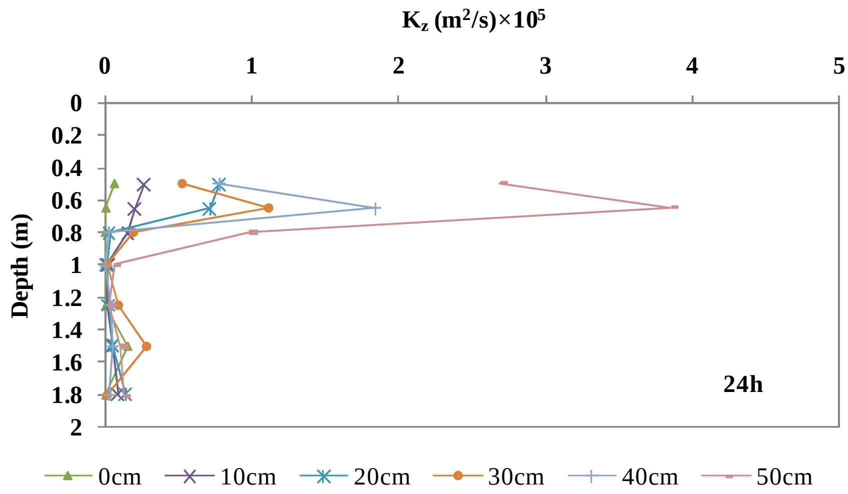


Figure 6. 8 Vertical diffusion coefficients at different diffuser depths

between 0.8 and 1.25 m. This was the bottom layer circulation and was not a factor to disturb stratification in the top layer. And in the surface layer recorded high values of diffusivity denote an intense turbulent activity, while the strong stratification in the lower layer (0.8 to 1.25 m) suppresses turbulence and limits the vertical exchange leading to smaller values of vertical diffusivity. Therefore, the effect of diffuser submergences on temperature profile was very low compared with cooling mixing from vertical diffusion coefficient profile studies.

Vertical diffusivity is a decreasing function of the buoyancy or Brunt-Vaisala frequency N^2,

$$N^2 = -\frac{g}{\rho}\frac{\partial \rho}{\partial z} \tag{5}$$

Where ρ is density of water, g is acceleration of gravity and z is depth.

All Brunt-Vaisala frequency values were calculated for Kz greater than 0. It indicated that all values were less than 0, and implied that the stratification was unstable in the upper layers. Therefore, from the discussion, the top 0.5m layer was assumed well-mixed and a strong gradient between 0.5 to 0.8 m for all conditions in this study.

Therefore, from the temperature profiles study, four layers were found. First top layer was a well mixed layer (0-50 cm) and the vertical diffusion coefficient was infinite in this layer. There was a strong gradient between 50 cm to 80 cm. Followed by the strong gradient layer, a weak diffusive layer was found between 80 cm to 1.25 m. This layer was very important to suppress turbulence and limit the vertical exchange to the lower layer. Below 1.25 m, a diffusion layer was observed. These four layers were presented in Figure 6.9.

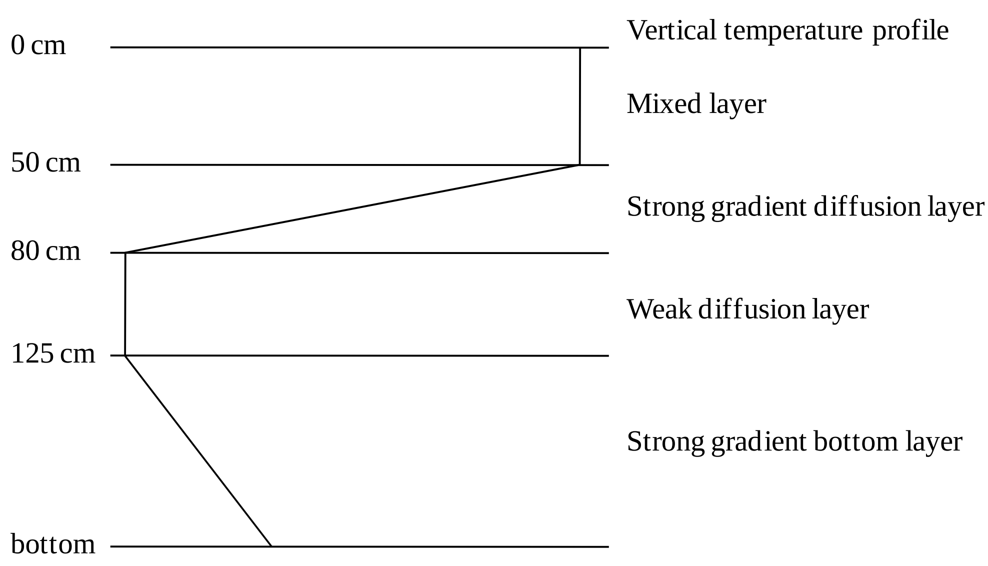


Figure 6. 9 Four layers observed in the study for temperature profiles

6.3.3 Effect of diffuser submergence on oxygen transfer efficiency

Submerged aeration systems are used to increase DO levels in water and promote water circulation. Submerged diffusers release air or pure oxygen bubbles at a depth and the ascending bubble plume entrains water, causing vertical circulation and lateral surface spreading. Oxygen transfers to the water across the bubble interfaces as the bubbles rise from the diffuser to the water surface. Oxygen transfer also occurs across the air-water interface at the free surface due to turbulence induced by bubble-plume motion and water circulation. In a surface aeration system, the influence of the air-water transfer rates and the bubble-water transfer rates at various depths in the water column can be important in designing diffuser placement. The relationship between diffuser depth and oxygen transfer is also an important design criterion for a surface aeration system with distributing diffusers. By understanding where oxygen transfer occurs, diffuser systems can be designed to either maximize bubble transfer at certain water depths or to maximize surface turbulence and mixing which induces greater free surface transfer (DeMoyer et al., 2003). In this part, the effect of surface aeration depths and circulation depths on aeration efficiency was studied and the direction of the holes in the diffuser (facing horizontal and up) was also considered as a factor to influence the oxygen transfer coefficient in the tank.

Effect of diffuser depths on oxygen transfer coefficient for diffusers either facing horizontal or up is presented in Figure 6.10. The curve in Figure 6.10 indicates that, if the effluent liquid flow rate is kept constant, the oxygen transfer coefficient is directly related to diffuser submergence (surface aeration depth) for surface aeration and circulation. For surface aeration with the diffuser facing horizontal, at 10cm submerges, the oxygen transfer coefficient was 0.357 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.357 h^{-1} to 0.275 h^{-1}, and then increased to about 0.413 h^{-1} at 40 cm submerges. However, the oxygen transfer coefficient was decreased to 0.311 h^{-1} at 50 cm submerges.

For surface aeration with the diffuser facing up, at 10cm submerges, the oxygen transfer coefficient was 0.334 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.334 h^{-1} to 0.284 h^{-1}, and then increased to about 0.3853 h^{-1} at 40 cm submerges. However, the oxygen transfer coefficient was decreased to 0.322 h^{-1} at 50 cm submerges. For surface circulation with the diffuser facing horizontal, at 10cm submerges, the oxygen transfer coefficient was 0.011 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.011 h^{-1} to 0.006 h^{-1}, and then increased to about 0.014 h^{-1} at 40 cm submerges, but decreased again to 0.009 h^{-1} at depth of 50 cm.

For surface circulation with the diffuser facing up, at 10cm submerges, the oxygen transfer coefficient was 0.039 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.039 h^{-1} to 0.026 h^{-1}, and then increased to about 0.042 h^{-1} at 40 cm submerges. However, the oxygen transfer coefficient was decreased to 0.016 h^{-1} at 50 cm submerges.

As shown in Figure 6.10, this phenomenon reflected the effect of surface aeration depth on oxygen transfer efficiency. For all four conditions, the greatest value of oxygen transfer coefficient was observed at 40 cm diffuser submergence. Moreover, the diffuser holes direction had no effect on the oxygen transfer coefficient for surface aeration, but the oxygen transfer coefficient of the diffuser facing up for surface circulation was greater than that facing horizontal (around 3.5 fold). It implied that surface aeration can add more oxygen into the tank except the natural aeration (surface turbulence) and the aeration efficiency of surface aeration was 10-fold greater than that of surface circulation.

For a given airflow rate and a given tank surface area, the oxygen transfer coefficient drops with the water depth, due to the higher bubble coalescence (Zlokarnik, 1979). Wagner et al. (1998) found that the specific oxygen absorption was reduced at greater depth of submergence where the depth ranged from 3.5 to 12 m in that study. Besides, Gillot et al. (2005) used dimensional analysis and found that for a given airflow rate and a given tank surface area, the oxygen transfer coefficient decreased with the water depth (submergence of the diffuser) for depth ranging from 2.2 to 5.9 m. Similar results also were recorded by other researchers (Khudenko et al. 1986; Al-Ahmady, 2006). The results reported herein indicated that, when the surface aeration depth was greater than 40 cm, the oxygen transfer coefficient was a decreasing function of the surface aeration depth, which was similar to those reported by other researchers. However, in this study, only the effect of surface aeration depths ranging from 10 cm to 50 cm on oxygen transfer coefficient were investigated and the oxygen transfer coefficient of shallow diffuser submergence might be dependent on the partial pressure of oxygen, contact time, temperature and driving force., which, unfortunately, was not researched in this study. Therefore, more study is needed to prove why 40 cm surface aeration depth can obtain best oxygen transfer efficiency in a venturi surface aeration system.

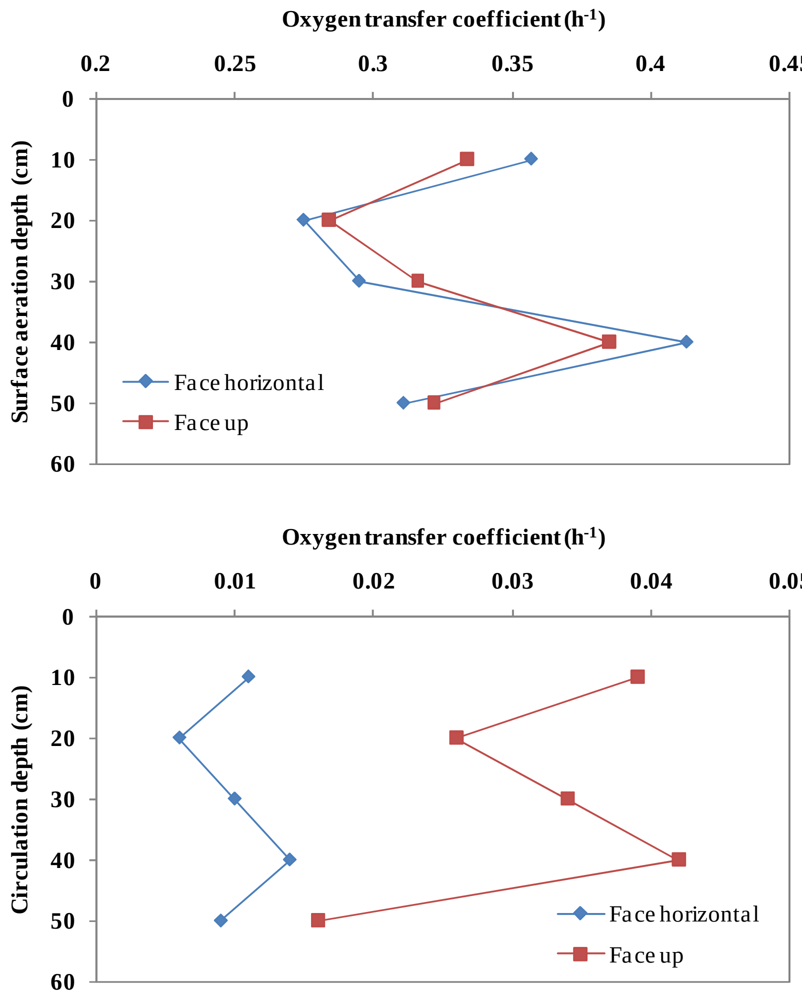


Figure 6. 10 Effect of surface aeration depth on the oxygen transfer coefficient of the system at different experimental conditions

6.3.4 Effect of surface aeration depth on oxygenation efficiency

The performance of aerators is evaluated by two parameters: 1) the standard oxygen transfer rate (SOTR) and 2) the standard oxygenation efficiency (SOE). The oxygen transfer rate is used to determine the number of aerators needed to meet the oxygen supply requirement for a particular aeration application. The oxygenation efficiency is used to estimate the energy requirement for operating the aerators. In wastewater treatment systems, oxygen transfer rate can be affected by several factors: 1) surface area of water in contact with air or oxygen; 2) mixing or turbulence; 3) saturation deficit (difference between saturated oxygen concentration and actual dissolved oxygen concentration); 4) influence of solids or other constituents in the water; and 5) temperature (Westerman and Zhang, 1997). All of these factors increase SOTR except for factor 4 (increasing solids loading conditions decrease SOTR). More information on SOTR and SOE was presented in Chapter 3.

The surface aerated layer may completely mix and remain well oxygenated throughout under surface aeration. Besides, vertical diffusion may occur in the surface aeration system. There were four layers obtained in our experiments, i.e., a well aerated layer (above 50 cm) and a strong gradient layer (50 cm-80 cm), and it also was found that diffuser submergence was not a dominant parameter to control the DO profiles in the surface aerated tank. From the studies on temperature, a 0.5 m well-mixed layer was determined for all the conditions. The effect of surface aeration depth on SOTR and SOE for surface aeration and circulation was presented in Figure 6.11.

From Figure 6.11, it can be seen that if the effluent liquid flow rate was kept constant, the changes of SOTR and SOE were similar to that of oxygen transfer coefficients as the saturation oxygen was not a factor to influence the values of SOTR and SOE for shallow surface aeration. The 40 cm aeration depth with the diffuser facing horizontal obtained the biggest value for both SOTR and SOE, which was thus considered the optimal surface aeration depth. However, in the field scale study of this project using surface aeration to reduce odor generation potential, a 30 cm surface aeration depth was used. It may therefore be inferred that better removal efficiency could be obtained if 40 cm surface aeration depth was employed.

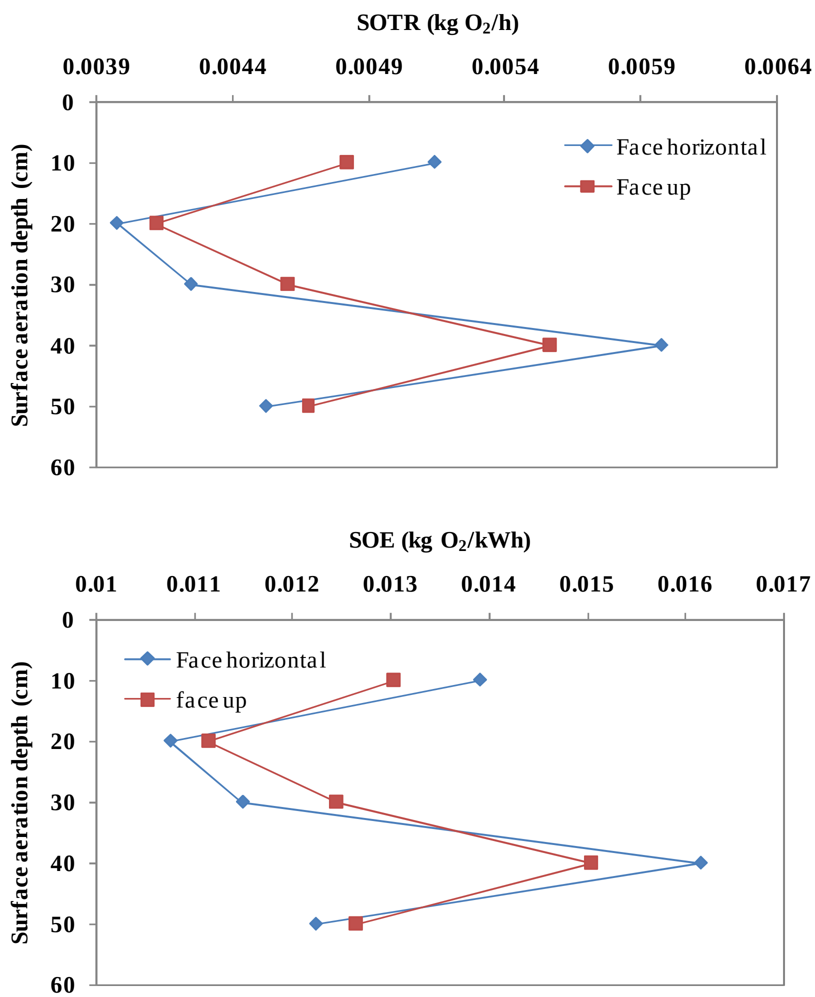


Figure 6. 11 Effect of surface aeration depth on the oxygen transfer coefficient of the system at constant effluent liquid flow rate

6.4 Conclusions

The effect of temperature and surface aeration depth on aeration efficiency in a surface aerated tank was studied to better understand some of the mixing processes involved and to provide data to obtain an optimal surface aeration depth. In order to get more parallel information, both surface aeration and surface circulation were investigated. The conclusions were presented as follows.

For 10, 20, 30, 40 50 cm surface aeration depth, deeper diffuser submergence didn't induce weaker stratification. Less than 1 m mixed layer was found for all surface aeration depths, which implied that this mixing was not caused by the diffuser system. Weather may have a much stronger influence on the stratification dynamics of the tank than diffuser mixing (forced convection).

The 50 cm surface aeration depth took the shortest time to obtain a uniform mixed 50 cm layer, while the 10 cm surface aeration depth took the longest. It implied that the surface aeration system can enhance the mixing above the diffuser but surface cooling and convective natural mixing and the tank temperatures may be more important.

Vertical diffusion coefficient values at upper layers (above 0.8m) ranged from approximately 0.002 to 0.160 cm^2/s, 0.004 to 0.146 cm^2/s, 0.003 to 0.388 cm^2/s, 0.009 to 0.692 cm^2/s, 0.002 to 0.510 cm^2/s, and 0.001 to 0.533 cm^2/s with an average 0.017, 0.023, 0.066, 0.120, 0.083 and 0.148 cm^2/s at surface aeration depth 0, 10, 20, 30, 40 and 50 cm, respectively. The highest vertical diffusion was found at 50cm surface aeration depth, while zero aeration had the lowest values.

All Brunt-Vaisala frequency values were calculated for K_z greater than 0. It indicated that all values were less than 0, and implied that the stratification was unstable in the upper layers. Therefore, from the discussion, the top 0.5m layer was assumed well-mixed and a strong gradient between 0.5 to 0.8 m for all conditions in this study.

For all four conditions, the greatest oxygen transfer coefficient was observed at 40 cm diffuser submergence. Moreover, the diffuser holes direction had no effect on the oxygen transfer coefficient for the surface aeration but the oxygen transfer coefficient of the diffuser facing up for surface circulation was greater than that facing horizontal (around 3.5 fold). It implied that surface aeration can add more oxygen into the tank except the natural aeration (surface turbulence) and the aeration efficiency of surface aeration was 10-fold greater than that of surface circulation.

The changes of SOTR and SOE were similar to that of oxygen transfer coefficients as the saturation oxygen was not a factor to influence the values of SOTR and SOE for shallow surface aeration. The 40 cm surface aeration depth with diffuser facing horizontal obtained the biggest value for both SOTR and SOE. Therefore, the optimal surface aeration depth of 40 cm was determined from this study.

Chapter 7 Summary and future research

7.1 Summary

This research was to understand the improvement of aerator modules, the effectiveness of lab-scale and field-scale surface aeration, factors influencing oxygen transfer efficiency, and the effect of temperature profiles and surface aeration depth. The following conclusions were drawn from this study:

7.1.1 Aerator improvement

Six configurations of venturi aerator modules were evaluated by determining their k_La, SOTR and SOE using the standard method of ASCE for testing aerators in clean water. For the series design (module a, b and c), oxygen transfer coefficients of 4.54, 3.79, and 3.58 h^{-1} were observed, respectively. While for the parallel design (module d, e and f), the corresponding values were 8.37, 5.93 and 11.87 h^{-1}. Module f achieved the highest oxygen transfer coefficient among all the aerator modules tested.

The SOTRs for the six aerator modules were determined to be 0.10, 0.09, 0.09, 0.18, 0.15, 0.31 kgO_2/h and the SOEs were 0.07, 0.06, 0.06, 0.12, 0.10, and 0.21 kgO_2/kWh, respectively. All the aerators tested showed lower SOTR and SOE than the published literature values for typical mechanical aerators. Again, module f achieved the highest performance among all the modules tested, whose SOTR increased by 3-fold and SOE by 3.5-fold compared to other five designs.

7.1.2 Lab-scale and field-scale surface aeration studies

Results obtained in this study showed that using the surface aeration system with venturi injectors can efficiently reduce odor generation potential at lab and field scales.

The intermittent surface aeration system at lab-scale is able to increase the oxygen concentration in the manure. The aeration treatment increased the solids removal efficiencies from 9.26% to 23.20% for TS and 15.53% to 45.78% for TVS, and decreased the ratio of TVS to TS from 0.57 to 0.35. The BOD removal efficiency remained stable around 90% to 95% after 3 to 4 weeks of aeration during which it increased linearly from about 7.5% to 90%. The VFAs in the aerated manure became nearly undetectable after 13 weeks of aeration. The VFA removal efficiency increased from 68.90% after one week to 89.1% (one month later) and around 95% (three month later). According to the results, a treatment time of 3 to 4 weeks will be needed to stabilize the liquid manure in order to maintain the VFA level below 230 mg/L.

Data showed that the surface aeration system studied at field-scale could achieve removal efficiencies of 39.64% and 16.55% for TVS and TS, respectively, after 4 months of continuous operation. The ratio of TVS to TS decreased from about 49% at the beginning to 31% at the end of the experimental period, while no significant difference was found between the treatment and the control. In the first 7 weeks, the BOD removal efficiency was lower than 20% (9% on average) and, afterwards, increased linearly from 21.5% to 86.5%. In contrast, the VFAs removal efficiency steadily increased from 5% to 85% and the levels of VFAs in the aerated liquid manure reached < 230 mg/L after 4 months of treatment. A linear relationship was found between BOD and VFAs with a correlation coefficient of 0.87 and the determination of correlation of around 0.76.

7.1.3 Factors influence aeration efficiency

The effect of aerator design, effluent liquid flow rate, alpha factor and surface aeration depth on oxygen transfer efficiency in a surface aeration system with venturi injectors was studied with conclusions achieved below.

It implied that the higher the effluent liquid flow rate, the better the oxygen transfer efficiency were for all tested aerator module designs. Based on the value of k_La, a 1.36-, 1.40-, and 1.51-fold increase in oxygenation capacity can be obtained from 2.9 L/s to 3.6 L/s for module a, b and c, respectively.

The alpha factor is a decreasing function of the equivalent contact time between bubbles and the liquid phase. Meanwhile, at a certain liquid flow rate, there is no big difference between two different modules at different effluent liquid flow rates.

Oxygen transfer coefficient is directly related to diffuser submergence. At 10cm depth, the oxygen transfer coefficient was 0.357 h^{-1}. With increasing the depth, the oxygen transfer coefficient was decreased from 0.357 h^{-1} to 0.275 h^{-1}, and then increased to about 0.413 h^{-1} at 40 cm. However, it decreased again to 0.311 h^{-1} at 50 cm depth.

7.1.4 Temperature and surface aeration depth

The temperature and surface aeration depth on aeration efficiency in a surface aerated tank were studied. For 10, 20, 30, 40 50 cm surface aeration depth, the deeper diffuser submergence didn't induce weaker stratification. The 50 cm surface aeration depth needed shortest time to obtain a uniform mixed 50 cm layer, while the 10 cm surface aeration depth took longest time.

Vertical diffusion coefficient values at upper layers (0.15m, 0.5m and 0.65m) ranged from approximately 0.000 to 0.238 cm^2/s, 0.009 to 0.552 cm^2/s, 0.002 to 2.936 cm^2/s, 0.006 to 0.090 cm^2/s, 0.004 to 2.711 cm^2/s, and 0.000 to 0.104 cm^2/s with an average 0.021, 0.122, 0.533, 0.032, 0.553 and 0.046 cm^2/s at surface aeration depth 0, 10, 20, 30, 40 and 50 cm, respectively. The highest vertical diffusion was found at 40cm surface aeration depth, while no aeration had the lowest values.

For all four conditions, the greatest value of oxygen transfer coefficient was observed at 40 cm diffuser submergence. Moreover, the diffuser holes direction had no effect on the oxygen transfer coefficient for the surface aeration but the oxygen transfer coefficient of the diffuser facing up for surface circulation was greater than that facing horizontal (around 3.5 fold). The 40 cm surface aeration depth with diffuser facing horizontal obtained the biggest value for both SOTR and SOE. Therefore, the optimal surface aeration depth of 40 cm was determined from this study.

7.2 Future research

In most research projects, the completion of one project generates even more questions and ideas for future research. Several research opportunities are described below, which stem from parts of the research completed in this thesis or ideas that come up during this thesis research.

More aerator module development and research is needed to improve the oxygen transfer efficiency of such aeration systems so effective control of odor from liquid swine manure lagoons can be achieved.

A field-scale surface aeration study to reduce odor generation potential with a 40 cm surface aeration depth can be conducted in order to prove whether it is an optimal surface aeration depth to gain better aeration efficiency. A bigger pump and smaller surface area should be considered in order to obtain better efficiencies in odor generation potential removal. Besides, a vertical profile of water parameters in the lagoon is needed.

More research on the effect of surface aeration related variables (bubble size, pressure, diffuser submergence, temperature, liquid flow rate and diffuser density et al.) is needed to better understand how these factors influence the aeration efficiency.

Based on the lab-scale experiments, study on temperature stratification at field-scale considering environmental conditions (solar radiation, wind energy, light attenuation, weather) is very important to determine actual surface aerated layer in a real lagoon using different surface aeration depths.

Based on the field-scale study, the temperature and dissolved oxygen models can be improved especially for this surface aeration system with venturi injectors.

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