Evaluation of a Wind Tunnel for Measurement of Odour Emission from Area Sources

By

Wenyue Wu

A thesis
Submitted to the Faculty of Graduate Studies
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A Thesis/Practicum submitted to the Faculty of Graduate Studies of The University of Manitoba in partial fulfillment of the requirement of the degree

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ABSTRACT

The wind tunnel method is commonly used for sampling odour from area emission sources. The performance of a wind tunnel in odour sampling is influenced by its design and aerodynamic characteristics. A laboratory study was conducted to evaluate the performance of a wind tunnel in odour sampling from a liquid emission surface. The wind tunnel evaluated consisted of an inlet bucket, a carbon filter, a fan, a PVC duct, an expansion section, a flux hood, a contraction section, a mixing chamber and two gas sampling ports. A perforated baffle was installed between the hood and the contraction section to facilitate creating uniform air flow in the hood. N-butanol solutions of various concentrations were used to simulate odour emission from liquid surfaces. An identical tunnel hood, which is 800 mm long, 400 mm wide and 250 mm high, was built with Plexiglas in order to perform velocity measurement and visual smoke testing. A total of 24 sampling ports were drilled for velocity measurement at the top surface of the hood at 200 mm, 400 mm and 600 mm of the hood length, 8 ports each length. The air velocity profiles were measured at bulk velocities of 0.20, 0.24, 0.30 and 0.38 m/s in the wind tunnel by inserting a hot wire anemometer at five different depth of each sampling port. The velocity profiles were deduced from the average velocities in the vertical and horizontal direction at 0.20, 0.24, 0.30 and 0.38 m/s bulk velocities. Smoke tests were conducted to visualize the air flow patterns in the tunnel. The flow pattern was recorded by a high speed motion camera and reviewed frame by frame. Computational Fluid Dynamics (CFD) modeling was performed to verify the flow patterns. The odour pick-up (recovery) rate was calculated from the mass balance of n-butanol in the solution and in the gas. N-butanol solution concentration was measured by using a spectrophotometer

together with an ethanol UV testing kit and the gaseous concentration at the exit of the wind tunnel was measured by using a photo ionization detector.

The measured velocity profiles exhibited a substantial degree of non-uniformity in the hood while the velocity at the central hood was relatively uniform. The lower the bulk velocity, the less the velocity variation. The air flow inside the hood was turbulent, confirmed by the calculated Reynolds Number of 4820 at the lowest bulk velocity of 0.20 m/s in the experiment. The pick-up rate ranged from 37% to 56% using the current sampling system. Therefore, caution should be exercised when interpreting and using odour emission data obtained with wind tunnels. Even similar wind tunnel designs could yield different performance results. The odour emission rate measured by the wind tunnel increased with the bulk air velocity in the tunnel and the relationship could be described by the power function.

KEYWORDS. Odour emission, Wind tunnel, Velocity profile, Pick-up rate

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TABLE OF CONTENTS

ABSTRACT..... i

ACKNOWLEDGEMENTS..... iii

LIST OF TABLES..... vii

LIST OF FIGURES..... viii

LIST OF APPENDICES..... xi

1. INTRODUCTION..... 1

2. OBJECTIVES..... 5

3. LITERATURE REVIEW..... 6

3.1 Introduction..... 6

3.2 Emission sources..... 6

3.2.1 Point source..... 6

3.2.2 Area source..... 7

3.3 Odur emission data from area sources..... 8

3.4 Odour emission rate measurement methods..... 9

3.4.1 Micrometeorogical methods..... 10

3.4.2 Flux chambers..... 10

3.4.2.1 Static flux chamber..... 11

3.4.2.2 Dynamic flux chamber..... 12

3.4.3 Wind tunnels..... 14

3.5 Wind tunnel types..... 15

3.5.1 Lindvall hood..... 15

3.5.2 Lockyer..... 16

3.5.3 Braschkat..... 16

3.5.4 Smith & Watts..... 16

3.5.5 UNSW tunnel..... 17

3.6 Aerodynamic performance of wind tunnels..... 18

3.7 Relationship between odour emission rate and air velocity inside the wind tunnel..... 19

4. MATERIALS AND METHODS..... 21

4.1 Description of the wind tunnel test system..... 21

4.2 Odour emission surface..... 22

4.3 Bulk velocity determination..... 23

4.4 Experiment procedure..... 25

4.4.1 Measurement of velocity profile..... 25

4.4.2 Smoke test..... 27

4.4.3 Computational Fluid Dynamics (CFD) modeling (Fluent)..... 27

4.4.4 Odour Pick-up rate evaluation..... 28

5. RESULTS AND DISCUSSION..... 31

5.1 Aerodynamics inside the wind tunnel..... 31

5.1.1 Velocity profile..... 31

5.1.2 Smoke test..... 38

5.1.3 Reynolds number calculation..... 39

5.1.4 CFD modeling..... 39

5.2 Pick-up rate (Recovery Efficiency)..... 45

5.3 Odour emission rate..... 48

6. CONCLUSIONS..... 52

7. RECOMMENDATIONS FOR FURTHER RESEACH..... 53

8. REFERENCES..... 57

LIST OF TABLES

Table 1	The wind profile parameter p as related to Pasquill stability classes	24
Table 2	Pasquil Stability Classes	25
Table 3	Coefficients of variation (CV) of measured velocities on vertical and horizontal planes at various locations in the wind tunnel for bulk velocities (BV) of 0.20, 0.24, 0.30 and 0.38 m/s	31

LIST OF FIGURES

Figure 1	Schematic drawing of experimental set-up	22
Figure 2	Wind tunnel hood dimension and locations of sampling ports	26
Figure 3	Lineal relationship between n-butanol concentration and absorbance difference for both calculated concentration and known concentration	30
Figure 4	The velocity profiles in the vertical direction at bulk velocity of 0.3 m/s	34
Figure 5	The velocity profiles in the horizontal direction at bulk velocity of 0.3 m/s	34
Figure 6	The velocity profiles in the vertical direction at bulk velocity of 0.2 m/s	35
Figure 7	The velocity profiles in the horizontal direction at bulk velocity of 0.2 m/s	35
Figure 8	The velocity profiles in the vertical direction at bulk velocity of 0.24 m/s	36
Figure 9	The velocity profiles in the horizontal direction at bulk velocity of 0.24 m/s	36
Figure 10	The velocity profiles in the vertical direction at bulk velocity of 0.38 m/s	37
Figure 11	The velocity profiles in the horizontal direction at bulk velocity of 0.38 m/s	37

Figure 12 Smoke flow pattern. (a) top view, (b) front view 38

Figure 13 CFD simulations of velocity profile at bulk velocity of 0.30m/s (front view) 41

Figure 14 CFD simulations of velocity profile at bulk velocity of 0.30m/s (top view) 41

Figure 15 CFD simulations of velocity profile at bulk velocity of 0.20m/s (front view) 42

Figure 16 CFD simulations of velocity profile at bulk velocity of 0.20m/s (top view) 42

Figure 17 CFD simulations of velocity profile at bulk velocity of 0.24m/s (front view) 43

Figure 18 CFD simulations of velocity profile at bulk velocity of 0.24m/s (top view) 43

Figure 19 CFD simulations of velocity profile at bulk velocity of 0.38m/s (front view) 44

Figure 20 CFD simulations of velocity profile at bulk velocity of 0.38m/s (top view) 44

Figure 21 Pick-up rates at bulk velocities of 0.20, 0.24, 0.30, 0.38 m/s and different n-butanol solution concentrations 47

Figure 22 Relationship between odour emission rate and bulk velocities derived from n-butanol release from the solution 51

Figure 23 Relationship between odour emission rate and bulk velocities derived from the gaseous n-butanol measurement at the exit 51

Figure 24 Schematic drawing of an idea wind tunnel 54

Figure 25 Schematic drawing of flat vanes installation 55

Figure 26 Schematic drawing of longer inlet pipe 55

Figure 27 Schematic representation of the quadratic distribution of samples points along a branch and its related area 56

LIST OF APPENDICES

Appendix I	Velocity Measurement Data.....	67
Appendix II	Bio-teck data recording.....	69

1. INTRODUCTION

Odours are generally ranked as the major generator of public complaints on air quality to regulatory agencies in North American communities. It has been estimated that about 70% of all complaints on air quality are related to odour (Watts and Sweeten, 1995). In Australia, Basarin and Cook (1982) reported that most of complains to environmental control agencies in Victoria concerned undesirable odours. The results of their survey showed that up to 91.3% of the respondents perceived air pollution as odours, 61.9% of them were annoyed by odours, and 28.6% of them were irritated by odours (Jiang et al., 1995). For the UK, the number of complaints about agricultural odours received by Local Authorities increased from 660 in the ten years period between 1960 and 1970 to 2478 in 1975 (Jones, 1977) and 3828 between April 1980 and March 1981 (Battersby, 1981). During 1989 and 1990 there were 3700 complaints about odour from farms, which is about 25% of all complaints received by the Environmental Health Officers (Skinner et al., 1997).

The number of odour complaints associated with animal agriculture however has declined recently in North America. The adoption of manure injection techniques for land application may be one reason for the decrease. Land application used to be the major cause of odour complaints when the incorporation and injection was not commonly practiced (Zhang, et al., 2002). Chen et al., 2001 reported that the manure injection resulted in low ammonia and odour concentrations immediately after manure application. Other reasons include less frequent manure land application due to increased manure storage capacity, covered manure lagoon, cleaner livestock housing and advanced manure handling system in the livestock facilities.

To control odour nuisances, odour must first be quantified. The odour emission rate from a source is of paramount importance for odour quantification and it provides a means of ranking odour sources, of assessing their impact on receptors through dispersion modeling, and is an essential design input for odour control schemes (Gostelow et al., 2003).

Typically, there are three types of odour emission sources: point, volume, and area sources. Point sources are where the odours are emitted from a single point, such as a stack. They are the easiest sources to measure, as the emission is confined within a flow channel, and representative concentrations and flow rates can easily be determined. Volume sources are typically buildings where odorous air escapes either intentionally through ducts or unintentionally through doors, windows or other openings. Emission measurement is more complicated here as the emission points are less well defined and controlled (Gostelow et al., 2003). Area sources are emissions from relatively large surface areas. Examples are landfill sites, wastewater tank/lagoon surfaces, or slurries applied to farmland.

Most odour emission sources associated with agriculture are point and area sources, such as exhaust from livestock buildings, manure storage, and manure applied land. Area sources present particular difficulties in odour measurement because, unlike point sources, there is usually no well-defined airflow associated with them. Area source emission may also vary substantially both spatially and temporally, which is affected by factors like temperature, wind and material transformation in the area source.

The determination of area source emissions can be carried out by using micrometeorological methods (Wilson et al., 1982; Wilson et al., 1983), the flux chamber

method (Denmead et al., 1980; Gulovsen et al., 1992) and the wind tunnel method (Lindvall, 1970; Lockyer, 1984; Ryden et al., 1985). Wind tunnel techniques have been identified as the best available method for the sampling of odour emission from area sources (Smith and Watts, 1994; Jiang and Kaye, 1996). Since Lindvall (1970) used wind tunnel techniques to compare the strength of odour from different area odour emission sources, a range of wind tunnels have been developed for estimating gaseous emissions (Lindvall et al., 1974; Lockyer, 1984; Braschkat et al., 1993; Smith and Watts, 1994b; Jiang et al., 1995).

For a wind tunnel system to be useful, it is necessary to have a thorough understanding of its aerodynamic performance (Gostelow, 2003). The basic assumption for estimating emission with a wind tunnel is that the airflow is completely mixed downwind of the emission chamber of the tunnel. However, early examples of wind tunnel systems were found to have highly non-uniform velocity and concentration profiles which can lead to errors in emission rate estimation. For example, Van Belois and Anzion (1992) experimented with acetone emission measurement using two wind tunnel systems, similar to the Lindvall et al., (1974) hood and the Lockyer (1984) wind tunnel. Velocity and concentration profiles were found to be non-uniform, leading to inaccuracies in obtaining representative velocity and concentration measurements.

Loubet et al., (1999a) evaluated the wind tunnel that was used for estimating ammonia volatilization from land by Lockyer (1984). They showed that the vertical profiles of wind velocity and gas concentration were non-uniform in the measurement section of the tunnel. The airflow was far from being completely mixed, leading to low recovery rates ranging from 77 to 87%.

Jiang et al., (1995) and Bliss (1995) developed the UNSW (University of New South Wales) wind tunnel from the original Lindvall (1974) hood. Experimentation with the Lindvall hood showed the velocity profile in the measurement section to be variable and non-uniform. A much improved velocity profile was achieved by extending the inlet duct, and fitting vanes and a baffle in the expansion section. Baldo (2000) established wind speed profile maps over the emission section for the UNSW wind tunnel (Jiang et al., 1995) and the Lockyer hood (Lockyer, 1984). Baldo (2000) indicated that many parameters affected the wind speed profile in the tunnels, including surface type, tunnel wind speed, entrance characteristic, wind tunnel shape and modifications to the tunnel geometry such as vanes and baffles.

There are numerous examples of portable wind tunnels used for odour emission measurement in the literature. However, there is a lack of consistency in the design and operation of these wind tunnels, and a lack of knowledge on the performance (accuracy, reliability, repeatability) of wind tunnels in odour sampling. This research, therefore, was conducted to evaluate a wind tunnel system similar to that that described by Jiang et al., (1995) for sampling odour from liquid emitting surfaces.

2. OBJECTIVES

Wind tunnels are commonly used for sampling odour from liquid emitting surfaces. Several research teams in North America have used wind tunnels similar to the design of the UNSW (University of New South Wales) wind tunnel to sample odour from manure storage. However, limited information is available on the performance (accuracy, reliability and repeatability) of these UNSW-like wind tunnels. The overall objective of this thesis was to evaluate a UNSW-like wind tunnel used at University of Manitoba to gain a better understanding of the wind tunnel performance as affected by design variations and operating conditions. The specific objectives of this project were broken down into the followings:

1. to examine air velocity profile inside the wind tunnel hood,
2. to verify the air flow behavior by using smoke testing and CFD (computational fluid dynamics) simulations,
3. to assess the pick-up (recovery) rate of n-butanol from a liquid emitting surface, and
4. to establish a relationship between the bulk velocity and measured odour emission rate.

3.0 LITERATURE REVIEW

3.1 Introduction

The determination of the odour mass flow emitted from a source is a very important step, and forms the basis for all subsequent considerations and calculations in odour assessment. Odour concentration is expressed as "odour units" (OU) (mostly in North America) or "odour units per cubic meter" (OU/m^3) (in Europe). One (1) European odour unit is defined as the amount of odorant(s) that, when evaporated into 1 cubic meter of neutral gas at standard conditions, elicits a physiological response from a panel equivalent to that elicited by one European Reference Odour Mass (EROM), evaporated into one cubic meter of neutral gas at standard conditions (CEN, 2003). One EROM is equal to 123 μg n-butanol (CAS 71-36-3) evaporated in 1 cubic meter of neutral gas at standard conditions. In the field of air pollution control, the pollutant concentration is commonly expressed as mass per unit volume (kg/ m^3). Therefore, the unit OU/m^3 seems logical to use for expressing odour concentration. Odour emission rate is the product of the concentration and the volumetric flow rate (m^3/s) of the air. Therefore, the unit of emission rate for point sources is OU /s, and for area sources $OU/s\text{-}m^2$.

3.2 Emission sources

3.2.1 Point sources

Point sources are controlled points at which ducted odours escape to atmosphere, usually through a stack (Gostelow et al., 2003). Emission rate measurements are relatively easy for these sources because confined airflow exists. However, the average flow rate should be measured with caution because velocity profiles within ducts are not

uniform. Odour samples are obtained by inserting a sampling probe into the stack and withdrawing the sample air from the airflow. Pre-dilution of moist, warm samples may be required to prevent formation of condensation in sample bags. The flow rate (the product of velocity and cross-sectional area) is usually measured using anemometers or pitot tubes at several positions across the stack cross-section. Measurement of flow rate in ducts is a mature technique and is covered in ISO 10780 (ISO, 1994).

3.2.2 Area sources

Area sources emit odour from a specified area. Examples of area sources include livestock feedlots, lagoons, land application of manure, waste treatment ponds, livestock buildings, landfill sites, etc. Emissions from area sources are usually governed by diffusion processes, whereby a concentration gradient provides the driving force for the transfer of odorants from the soil or liquid to the air. In some cases, there may also be a significant convective element, whereby a pressure gradient provides an additional driving force. This often occurs in landfill sites, where gas is generated by the biochemical degradation of the filled waste.

Measurement of these emissions poses some real difficulties. First, there is no well-defined airflow associated with the area source. Unlike point sources, for which airflow could be easily sampled and measured from the exit chimney or exhaust fan and emission rate could be calculated by multiplying the measured concentration by the flow rate. Second, the emission rate most likely varies considerably with time, over short times due to such factors as temperature, radiation, surface moisture content, wind speed and over longer periods due to the growth or decline of the emission generation factors.

Finally, emissions vary spatially due to the different spatial histories of the above mentioned variables.

3.3 Odour emission data for area sources

In Australia, the odour emission data used in regulating feedlots have been adapted from two studies (Smith and Watts, 1994a). The first study was undertaken at a 20,000 head feedlot near Peechelbah in Victoria and the second at Cannon Hill in Brisbane, Queensland. Neither of these studies was comprehensive and adequately reported in the scientific literature. Nevertheless, the data have been used in numerous environmental impact assessments, in the preparation of feedlot guidelines and in several court actions. Using these data, the Queenland feedlot guidelines were developed in which it is assumed that the rate of odour emissions from a dry feedlot with a stocking density of 10 m^2/head would be 0.5 OU/s-m^2.

In North America, much research was conducted to study the odour emissions from agricultural sources. According to Hartugn and Phillips (1993), the majority of gaseous emissions in animal housing come from stored manure. Stored manure and urine produces 40 different types of gaseous emissions when undergoing anaerobic decomposition (Priest et al., 1994).

Emission rates found in the literature are extremely variable. Frederic et al. (2004) reported the odour emission from the manure tank was 12 OU/s-m^2. Byler et al. (2004) studied three phototrophic and three non-phototrophic anaerobic swine lagoons in eastern Nebraska from late spring to summer. They reported the odour emission rates from phototrophic lagoons were relatively constant from late spring to summer (from 4.0 to

24.5 OU/s-m^2). Gay et al. (2003) reported the odour emission rate from concrete manure storage tank was equal to 30.1 OU/s-m^2. The average results obtained from single, 1st and 2nd cell earthen basins were 14.4, 22.1 and 12.0 OU/s-m^2, respectively. Lague et al., (2003) reported odour emission from manure storage facilities varied from 1.1 to 8.7 OU/s-m^2. Lim et al. (2003) measured odour emissions from two 1st cell earthen basins located in Illinois and Indiana. The average result obtained was 1.5 OU/s-m^2. Bicudo et al. (2003) obtained results varying between 6 and 18 OU/s-m^2 from earthen basin in Minnesota. Jacobson et al. (1999) and Schmidt et al. (1999) also measured odour emissions from earthen basins and obtained results varying between 2.2 and 18 OU/s-m^2, respectively.

Odour emissions vary widely during the day as well. Watts et al. (1992) reported that in 2 days immediately following heavy rainfall, emissions increased steadily to values in excess of 800 OU/s-m^2 then declined equally rapidly (as the pad dried) to around 45 OU/s-m^2 by the end of the fourth day.

3.4 Odour emission rate measurement methods

Three techniques are described by Wilson et al., (1982) for measuring emissions from area sources: simulation testing, direct measurement and indirect measurement. Simulation testing is not in use now because of its limitation and disadvantages. Indirect measurement includes micrometeorological methods using different atmospheric dispersion models. Direct measurement uses enclosures (hoods) to cover a small area of emitting surface, and the odour concentration in the enclosed space is measured. The direct methods include the static flux chambers, dynamic flux chambers and wind tunnels.

3.4.1 Micrometeorological methods

Micrometeorological methods give an indirect measurement of emission rate by sampling wind velocity and odour concentration downwind of the emission source (Gustolow et al., 2003). Sampling may occur at various heights across the entire plume section, or may employ models of spatial concentration within the plume to minimize the number of measurements required. These methods are generally more suited to analytical rather than sensory odour measurements, because dispersion will often result in low odour concentration downwind of the source, and sensory measurements may lack the sensitivity necessary to differentiate plume concentrations from background odours. In addition, some methods require a large number of concentration samples, which adds to cost and causes practical problems if olfactometry is used. Where emission rates vary spatially, employing a micrometeorological technique can be advantageous as these can measure the total emissions from a large site. Unfortunately, micrometeorological technique can not be applied to all sites, particularly where emission rates are very low, or where there are upwind odour sources which may interfere with concentration measurements.

3.4.2 Flux chamber methods

For flux chamber (hood) techniques, as many locations in as short a time period as possible should be sampled in order that the hood measurements can be reliably extrapolated to give site-wide flux values (Mosher et al., 1999). This means the hood technique used must be very portable and capable of making rapid flux measurements. Mosher et al., (1999) found static flux chambers to be useful for this purpose: a team of

three individuals were able to conduct 60-100 flux measurements per day on a landfill site.

3.4.2.1 Static flux chamber

The basis of the static flux (or equilibrium) chamber is to seal a portion of the emission surface and to allow the gas concentration to reach equilibrium within the chamber. The emission rate is then determined by the change in emission concentration over time using equation 1 (McGinn and Janzen, 1998):

$$N = \frac{V_c}{A_s \Delta t}\left[C_{t_2} - C_{t_1}\right] \quad (1)$$

where:

N = emission rate

V_c = chamber volume

A_s = enclosed surface area

Δt = time interval (t_2-t_1) over which concentration C_{t1} and C_{t2} are measured

C_{t1}, C_{t2} = odour concentration at time t_1, t_2

Due to very low velocities in these chambers, there is no enhancement of emission rate by reduction of boundary layers. In addition, emission rates will reduce over time and approach zero as equilibrium concentrations are reached, assuming the sample airflow is small compared to the chamber volume. These chambers are likely to predict very low emission rates as a result.

Tillman et al., (2002) compared static flux chamber results against modeled results for a soil test-cell. Measured and modeled fluxes were found to be in good

agreement for diffusive and diffusive-convective emissions. This flux chamber utilized recirculation of sampling air, and employed adsorbent traps for collection of samples, so it would have had relatively good mixing compared to a completely static chamber and may also have been less susceptible to suppression of emissions due to concentration build-up in the chamber. Perera et al., (2002) also performed tests on a static flux chamber on a soil test-cell. Emission rates measured with the flux chamber were found to underestimate actual flux rates by 15-65% due to increased concentrations within the chamber. The largest errors occurred for the largest fluxes and smallest chamber volumes. Mosher et al., (1999) applied static flux chambers to measurements of landfill gas emissions and found these to be in good agreement with micrometeorological measurements.

3.4.2.2 Dynamic flux chamber

Dynamic flux chambers are similar to static flux chambers but utilize controlled airflow as part of the emission rate measurement. The airflow used in dynamic flux chambers is low, typically in the range 1-40 L/min. The emission rate is determined as the product of the airflow through the chamber and the odour concentration.

The incorporation of airflow improves the emission rate measurement, but they are still susceptible to emission rate reduction due to high concentrations within the chamber if the airflow is low, and they may also be susceptible to poor mixing and dead zones (Gao and Yates, 1998; Zhang et al., 2002). Special arrangements for the introduction of the sweep flow into the chamber, such as circular flow rings, may promote good mixing (Cooper et al., 1992). Circulation fans may also be included within

the chamber to improve mixing. It is also important to allow steady-state conditions to be achieved. Initial concentrations in the chamber may be high if it is placed on the emission surface prior to allowing the sweep air to flow through; this could produce an artificially high emission rate measurement if not allowed to stabilize (Park and Shin, 2001)

As emissions have been shown to increase with airflow in some cases in dynamic flux chambers (Bowker et al., 1994; Cooper et al., 1992; Lindberg et al., 2002; Sadek et al., 1998), there is still a problem of selecting a suitable airflow for the emission measurement. Dynamic flux chambers pose problems as it is not possible to determine the flow velocity in the chamber; this makes it difficult to relate the flux chamber measurement to field conditions where wind speed may be an important determinant on the emission rate.

There are several examples of the use of dynamic flux chambers in the literature, for emissions from both liquid and soil surfaces. Adams et al., (1981) used dynamic flux chambers to determine biogenic sulphur emissions from soils. Park and Shin (2001) and Guy and Page (2001) applied dynamic flux chambers to measurement of landfill gas emissions. Frechen (1992) applied a flux chamber to measurements of odour emissions from sewage treatment processes.

3.4.3 Wind tunnel methods

Wind tunnels are portable, open–bottomed enclosures which are placed over the emitting surface (Smithand Watts, 1994a). Ambient or filtered air is blown or drawn through the tunnel to mix with and transport the emissions away from the emitting surface. The odorous air is sampled at the outlet of the tunnel and the odour emission rate

is estimated by multiplying the outlet stream odour concentration and the airflow rate through the tunnel. When the ambient air is used the concentrations of both the incoming and outgoing air streams are sampled and difference between the two is used in determining the emission rate.

The airflow rate in the wind tunnel is typically much higher than in a flux chamber and the wind direction in the tunnel is more defined than in a flux chamber (Schmidt and Bicudo, 2002).

Benefits of the wind tunnel over the traditional flux chamber are the larger surface area covered, and the air exchange rates or air speed in the tunnel being similar to ambient conditions. Smith and Watts (1994) and Jiang and Kaye (1996) described the wind tunnel as the best available method for the sampling of odour emission from area sources. The odour emission rate from area source is expressed as the product of the measured odour concentration and the flow rate as follows:

$$E = CV_t \frac{A_t}{A_s} \tag{2}$$

where:

E = odour emission rate (OU/s-m^2)

V_t = bulk wind speed in the tunnel (m/s)

A_t = cross-sectional area of the tunnel (m^2)

A_s = surface area covered by the tunnel (m^2).

A major disadvantage of wind-tunnel type hoods arises from the relatively large airflow used. These result in dilution of the emitted gases, so that wind tunnel methods can lack sensitivity and be unsuitable for very low emission rates. This is an area where

static and dynamic flux chamber are preferable, as the relatively low sweep airflow means there is minimal dilution of the emitted gases (Perera et al., 2002). For circumstances where emission rates are low and wind effects are minimal, such as when emission surfaces are shielded from the effects of the wind by vegetation, static or dynamic flux chambers can be a good method, provided that care is taken to minimize concentration gradient or pressure effects by using short sampling times (Cooper et al., 1992, Mosher et al., 1999).

3.5 Types of wind tunnel

3.5.1 Lindvall tunnel

There are numerous examples of portable wind tunnels which have been used for emission rate measurement in the literature. One of the earliest examples of a portable wind tunnel used to collect odour samples was described by Lindvall et al., (1974). The tunnel consisted of a rectangular measurement section, open to the emission surface, with contraction and expansion sections allowing for air to be blown through the measurement section. A tunnel similar to the Lindvall et al., (1974) was applied to odour and hydrogen sulphide emissions from manure by Schmidt and Bicudo (2000).

3.5.2 Lockyer tunnel

Another early wind tunnel system was described by Lockyer (1984), which was applied to measurement of ammonia volatilization from grass swards. The measurement section of the wind tunnel system was constructed of transparent polycarbonate to allow sunlight penetration. The air velocity in the wind tunnel was matched to the wind velocity

at 25 cm in height outside the tunnel, allowing the growing plants to be exposed to realistic wind velocities. Temperature differences inside and outside the tunnel were found to be small. Condensation problems were apparent on the walls of the tunnel, which was eventually prevented by using car windscreen type heating panels.

3.5.3 Braschkat tunnel

Braschkat et al., (1993) describes a large wind tunnel system which was applied to ammonia emission from soils by Mannheim et al., (1995). As in Lockyer's (1984) experiments, air velocity in the wind tunnel was matched to ambient wind speeds. A similar wind tunnel was applied to ammonia emissions from soils by Reitz and Kutzbach (1997).

3.5.4 Smith and Watts tunnel

Smith and Watts (1994b) experimented on 'large' and 'small' wind tunnels. The large tunnel was similar to the Lockyer (1984) tunnel. The tunnels were applied to ammonia emissions from feedlot pads. Results showed that the ammonia emission rate variations with wind velocity were similar for both tunnels, but differences were apparent in the emission rates: the smaller tunnel consistently gave higher emission rates. This was thought to be due to different velocity profiles within the wind tunnels, and the smaller tunnel had a higher velocity close to the emission surface.

3.5.5 UNSW wind tunnel

Jiang et al., (1995) attempted to design a wind tunnel that provided stable horizontal and vertical flow velocities throughout the tunnel. The resulting design was a tunnel 0.8 m long, 0.4 m wide, and 0.25 m high, similar to the Lindvall (1974) tunnel. A perforated baffle and wind vanes were installed in the inlet expansion chamber to create uniform airflow in the tunnel. Results from the initial studies showed a much improved velocity profile in the tunnel. Additional modifications were made to this wind tunnel to improve the mixing in the exhaust chamber (Wang et al., 2001). Wang et al., (2001) developed a special sampling chamber extension and a sampling manifold with optimally distributed sampling orifices for the wind-tunnel sampling system and the recovery efficiency of sampling was improved to 83-100% with an average of 90%.

The UNSW wind tunnel is most commonly used nowadays due to its developed airflow behavior. University of Minnesota built a wind tunnel similar to the UNSW wind tunnel to determine odour emissions from manure surfaces (Schmidt et al., 1999).There is another similar wind tunnel used at the University of Manitoba, which has longer inlet duct, bigger expansion and contraction sections and perforated baffle with denser holes, but possesses the same tunnel shape. The wind tunnel has been used to sample air from lagoons and for laboratory measurements for the past three years.

3.6 Aerodynamic performance of wind tunnels

For a wind tunnel system to be useful it is necessary to understand the aerodynamics of the tunnel. The aerodynamic performance of a wind tunnel is considered a critical parameter in ensuring the reliability of odour sampling (Jiang and Kaye, 2001).

An underlying assumption in using wind tunnels is that the airflow is completely mixed downwind of the emission chamber of the tunnel. However, the wind profile results from conventional type wind tunnels show strong crosswind and vertical gradients, highlighting the need for a careful analysis of the turbulence inside the tunnel (Van Belois and Anzion, 1992).

Loubet et al., (1999a) evaluated the wind tunnel of Lockyer (1984) that was used for estimating ammonia volatilization from land. They showed that the vertical profiles of wind velocity and gas concentration were non-uniform in the measuring section of the tunnel. The airflow was far from being completely mixed leading to a recovery rate ranging from 77 to 87%.

Baldo (2000) established a wind speed profile map over the emission section for the UNSW wind tunnel (Jiang et al., 1995) and the Lockyer hood (Lockyer, 1984). Baldo (2000) indicated that many parameters affected the wind speed profile in the tunnels, including surface type, tunnel wind speed, entrance characteristic, wind tunnel shape and modifications to the tunnel geometry such as vanes and baffles.

The UNSW tunnel developed by Jiang et al., 1995 represents a useful development of the Lindvall hood, and was reported by Jiang et al., (1995) to have relatively uniform horizontal and vertical flow velocities throughout the tunnel. Additional modifications were also made to this wind tunnel to improve the mixing in the exhaust chamber and sample from a cross sectional area in the final exhaust port (Wang et al., 2001). Several similar wind tunnels were built following the UNSW design (Smichdt, 1999; Zhang, 1999).

3.7 Relationship between odour emission rate and air velocity in wind tunnels

The basic principles governing the mass transfer from surfaces to fluid media suggest that emissions are dependent on velocity of the fluid medium near the surface (Schmidt and Bicudo, 2002). Other factors, such as tunnel geometry and materials used to construct the tunnel, are also expected to influence the results (Smith and Watts, 1994a). Smith and Watts (1994a) evaluated the performance of two wind tunnels of different sizes in measuring odour emission from feedlots and reported that there was strong dependence of the measured emission rate on the air velocity inside the tunnel.

Because emission rates can vary with wind speed, it is advantageous to develop a relationship between wind speed and emission rate, especially for liquid surfaces. This allows the use of variable emission rates in subsequent modeling. If emission rates are measured to serve as the input to dispersion models, it is essential that the relationship with wind speed be included if the model results are to be meaningful.

Selection of suitable wind speeds for wind tunnels could be difficult. In many European studies, a constant bulk tunnel wind speed of 1m/s was used. This seems to be at odds with the important influence of tunnel wind speed on emission rates observed by Pain et al., (1990), Ryden and Lockyer (1985), Homans (1987), and Bouwmeester and Vlek (1981). Lockyer (1984) and Mannheim et al., (1995) varied wind tunnel air velocities to match ambient wind speeds during the course of measurement. Jiang and Kaye (1997) determined the appropriate velocity for the UNSW wind tunnel on the basis of wind speed associated with the most odour complaints.

Wind tunnel derived emission rates are generally taken as a function of the average velocity across the tunnel section. It is more useful if the emission rate is taken as

a function of the velocity profile (Smith and Watts, 1994b; Leyris et al., 2000), which could be expressed in terms of shear stress, velocity profile or friction velocity. This allows wind-tunnel measured emission rates to be adjusted for different wind speeds using theoretical and empirical emission models.

In the study by Smith and Watts (1994b), odour emissions were measured to increase as bulk tunnel wind speed increased from 0.2 m/s to 2.0 m/s. From these measurements a power equation was developed to describe the relationship between the emission rate and the wind speed:

$$\frac{E_v}{E_1} = 1.05V^{0.63} \qquad R^2=0.685 \tag{3}$$

where:

V = given bulk tunnel velocity (m/s)

E_1 = the emission at 1m/s

E = emission at velocity V.

Results by Schmidt et al (1999) show the relationship for odour to be $E_v/E_1=V^{0.89}$ and $E_v/E_1=V^{0.90}$ for hydrogen sulfide with R^2 values of 0.56 and 0.87 respectively. Bliss et al., (1995) studied the effect of bulk wind speed on ammonia emissions and found the relationship to be a power function with an exponent of 0.5. Other researchers also found that odour emissions increased as tunnel wind speed increased. The exponent of the power function relating odour emission to the tunnel wind speed ranged from 0.4 to 1.2 (Bouwmeester and Vlek, 1981; Pain et al., 1990; Homans, 1987).

4. MATERIAL AND METHODS

4.1 Description of wind tunnel test system

The main components of the experimental system included a water tank and a wind tunnel (fig. 1). The 400-L water tank was filled with n-butanol solution to simulate an odour emitting surface. The wind tunnel was built by adopting a design from the University of New South Wales (UNSW), Australia (Jiang et al., 1995). The wind tunnel consisted of a carbon filter, a fan, a PVC duct, a hood and two gas sampling ports. The fan was powered by a 12 V DC source and a control unit (Lab-line Instruments, Inc., Melrose Park, IL) was utilized for fine adjustment of airflow rate. The fan pulled air through an activated carbon filter mounted at the inlet of the wind tunnel air duct to remove any odour that might exist in the incoming air. The hood, expansion and concentration sections, and the mixing chamber were constructed with stainless steel. The bottom of the chamber (hood section) was open to the emission surface, covering an area of 0.32 m^2 (0.8 × 0.4 m).

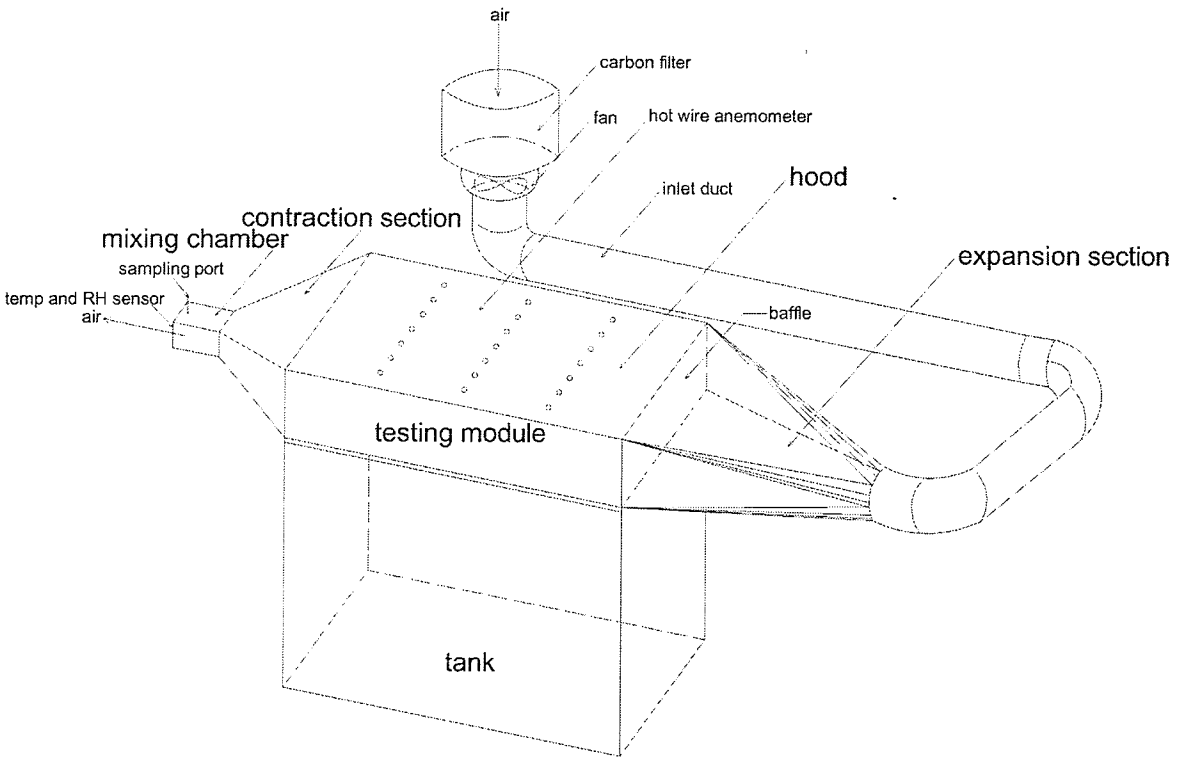


Figure 1. Schematic drawing of experimental set-up

4.2 Odour emission surface

Aqueous n-butanol ($CH_3CH_2CH_2CH_2OH$, 99.9%, F.W. 74.12) was used to make various solution concentrations to simulate odour emission from liquid surfaces. N-butanol was used because it is non-toxic, has good stability in air and water, and is used as a reference for odor intensity measurements (ASTM, 1997; CEN, 2003). N-butanol solutions were prepared with the following double geometric series of concentration: 500, 1000, 2000, and 4000 ppm. Each solution of n-butanol in the 400 L tank was mixed mechanically for 30 minutes by an electric stirrer (Model 6000, Arrow Engineering CO., Inc, Hillside, NJ) before placing the wind tunnel onto the tank. The solutions were stirred again for another 30 minutes before each bulk velocity measurement. During the stirring period, the tank was covered by a plastic sheet to avoid n-butanol volatilization. The 30

min stirring was necessary to create uniform n-butanol solution because n-butanol was floating on the water due to its lighter density. Take the 500 ppm solution for example, after 30 min stirring, the n-butanol solution concentrations were 437, 440 and 446 ppm for samples collected from the tank bottom, 30 cm below the surface and surface, respectively. The difference in concentrations between the bottom and surface was about 2%, indicating a good uniformity of the solution.

4.3 Bulk velocity determination

An important operating parameter of the wind tunnel is the average airflow velocity passing the hood, namely the bulk velocity. To maintain the air velocity similar to the ambient condition, the surface velocity at 0.125 m (half of the tunnel height) was considered. The relationship between wind velocity and elevation is described by equation 4 (De Nevers, 1995).

$$\frac{u}{u_1} = \left(\frac{z}{z_1}\right)^p \quad p: 0.07 - 0.60 \tag{4}$$

where:

u = velocity at z height (m/s)

u_1 = velocity at z_1 high (m/s)

z, z_1 = height (m)

p = exponent (Table 1)

Odour disperses farthest at the most stable atmosphere, The Pasquill stability classification defined in Table 2 is commonly used to describe the atmospheric stability (Wark et al., 1998). Class F is the most stable condition, and therefore, was chosen as the worst situation. This resulted in an exponent p-value of 0.55 (Table 1) in equation 4. In

Winnipeg, the average wind speed at 10 m height is 4.47 m/s (Winnipeg Weather Data, 1995-1999). At 125 mm above the ground, the wind speed should be $4.47 \times (0.01)^{0.55} = 0.35$ m/s.

Table 1: The wind profile parameter p as related to Pasquill stability classes (after Wark et al., 1998)

Pasquill Classes	Exponent p	
	Rural	Urban
A	0.07	0.15
B	0.07	0.15
C	0.10	0.20
D	0.15	0.25
E	0.35	0.30
F	0.55	0.30

However, for Class F, the wind velocity is between 2 to 3 m/s at 10 m height (Table 2). This would result in a velocity range of 0.16 to 0.24 m/s in the wind tunnel, as per equation 4.

Table 2: Pasquil Stability Classes

A: strongly unstable B: moderately unstable

C: Slightly unstable E: slightly stable

F: moderately stable D: neutral (overcast)

Surface Wind speed (at 10m) m/s	Day Solar radiation, W/m^2			Night	
	Strong >600	Moderate 300-600	Slight <300	Thinly overcast >4/8	Clear <3/8
0-2	A	A-B	B	-	-
2-3	A-B	B	C	E	F
3-5	B	B-C	C	D	E
5-6	C	C-D	D	D	D
>6	C	D	D	D	D

(Courtesy of Wark et al., 1998)

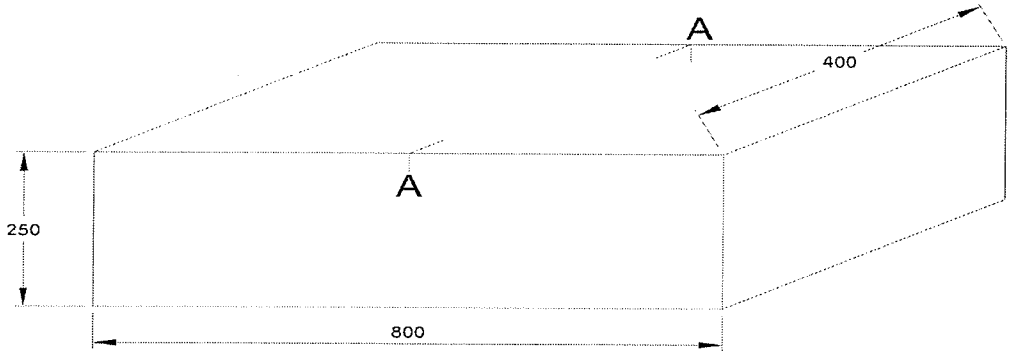
Jiang (1995) suggested 0.33 m/s was the optimum velocity for use of UNSW wind tunnel. Smidt (2000) considered 0.3 m/s as the most commonly used bulk velocity in UNSW-type wind tunnel research.

Based on the above discussion, bulk velocities of 0.20, 0.24, 0.3, 0.38 m/s were used in this study.

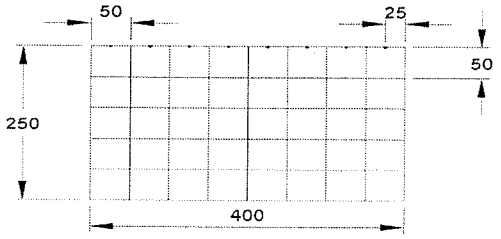
4.4 Experimental procedure

4.4.1 Measurement of velocity profile

An identical hood section (the main body of the wind tunnel) was made with Plexiglas to replace the stainless steel one for visualizing air flow (fig 2). The top surface was drilled three groups of sampling ports at ¼ (x = 200 mm), ½ (x = 400 mm) and ¾(x = 600 mm) of the hood length. Each group had 8 ports 50 mm apart and each port was 11 mm (7/16 in) in diameter. The sampling ports were sealed by rubber stoppers and only one port was opened at a time for taking measurements.



main section of wind tunnel(mm)



A-A section(mm)

Figure 2. Wind tunnel hood dimension and locations of sampling ports.

A hot wire anemometer (Llorite 800, Bacharach, Pittsburgh, PA) was used to measure velocities in the hood. At each sampling port, the air velocity was measured at

distances of 25, 75, 125, 175, and 225 mm above the solution surface. Thus, each of the three cross-sections had 40 velocity measurements. The horizontal velocity distribution was produced by averaging the velocities at 5 different heights for each horizontal level at each position. Similarly, for each height, the velocities at 8 different horizontal positions were averaged.

In addition, solution temperature was measured 100 mm below the solution surface using a digital thermocouple indicator and room temperature and humidity were recorded by using a temperature and relative humidity sensor (RH-20C, Omega Engineering, Inc., Stamford, CT).

4.4.2 Smoke test

The flow pattern inside the hood was investigated by observing the movement of a tracer – smoke. A smoke generator (Colt, Concept Engineering Ltd, Maidenhead, Berks, England) was connected to the wind tunnel intake. A high speed motion camera (MotionScope®PCI, Redlake MASD, In., San Diego, CA) was used to record the smoke patterns inside the hood. The recording rate was 125 frames per second and the images were saved on a computer hard drive for analysis.

4.4.3 Computational Fluid Dynamics (CFD) modeling (Fluent)

A commercial Computational Fluid Dynamics (CFD) modeling package, Fluent 6.2.16 (Fluent Inc., Lebanon, NH) was used in this study to simulate the air movement in the wind tunnel. The model geometry was created by Gambit 2.3.16 (Fluent Inc., Lebanon, NH) with 315189 nodes after meshing. The boundary zones included the inlet-

velocity, outlet-pressure, tunnel walls, and baffle-zone. The inlet-velocity was set at different pipe velocities of 2.54, 3.05, 3.81, 4.83 m/s and outlet-pressure was set at standard pressure of 101.3 kPa. In the modeling, the flow and turbulent equations were solved using under-relaxation factors at adiabatic condition.

4.4.4 Odour pick-up rate evaluation

The odour pick-up rate was evaluated based on mass conservation as shown in equation 5. The rate was computed as the amount of n-butanol measured in the gas divided by the mass loss from the solution in a 10 min interval. Use of Gas Chromatograph (GC) was attempted to measure gaseous n-butanol concentration. The GC results showed substantial degree of uncertainty. Therefore, the gaseous n-butanol concentration at the exit of the wind tunnel was measured by using a GasAlertMicro5 Photo Ionization Detector (PID) (BW Technologies LP, Calgary, AB). The PID sensor was calibrated against 50 ppm butyl alcohol-1 cylinder (BOC Group, Inc, Murray Hill, NJ). Prior to calibrating the PID sensor, the span concentration value was set as 50 ppm, matching the n-butanol gas used. The calibration continued for 30 seconds and the PID sensor attained a sufficient level of n-butanol gas.

$$P = \frac{C \times M \times v \times A_s \times t}{24.5 \times V_t \times \left(C_{t_1} - C_{t2}\right)} \tag{5}$$

where:

P = pick up rate (%)

C = gaseous n-butanol concentration (ppm)

M = n-butanol molecular weight (74.12)

v = bulk air velocity (m/s)

A_s = hood section area (0.1 m^2)

t = time interval (t_2-t_1) (600 s)

C_{t1}, C_{t2} = n-butanol solution concentration at time t_1 and t_2 (mg/L)

V_t = tank volume (400L)

N-butanol solution samples were collected at the solution surface and measured in the laboratory using the Universal Microplate Spectrophotometer (PowerWaveTMXS, BIO-TEK® INSTRUMENTS, Inc., Winooski, VT) together with Ethanol UV- method Testing Kit (Cat.No.10176290035, R-Biopharm AG, D-64293 Darmstadt, Germany). A reaction mixture of NAD/AI-DH tablet(s) and Potassium diphosphate buffer from the Testing Kit was made according to the number of testing samples and was dispersed into a 96 well assay plate. Blank and n-butanol samples were added into the wells using three replications. The absorption of the solutions (A_1) was read after thoroughly mixing for 3 minutes. Absorbance A_2 was also read after adding ADH suspension solution and after mixing 10 min. The n-butanol solution concentration was calculated based on absorbance difference (A_2-A_1) as follows (Manual for Ethanol UV-method Testing Kit).

$$C = \frac{V \times MW \times \Delta A}{\varepsilon \times d \times v \times 2 \times 1000} \tag{6}$$

where:

V = final volume (0.23 mL)

v = sample volume (0.02 mL)

MW = molecular weight of n-butanol (74.12 g/mol)

d = light path length (0.6 cm)

ε = extinction coefficient of NADH at 340 nm 6.3L/(mmol-cm)

ΔA = absorbance difference (A_2-A_1)

The accuracy of the Universal Microplate Spectrophotometer with Ethanol UV method was checked against pre-diluted n-butanol solutions with concentrations from 8.5 to 136 mg/L, which were shown from 0.17 to 2.72 mg/L per microplate well in the figure 3. The results revealed both "known" and "calculated" concentration were linear with absorbance difference (A_2-A_1) as shown in Figure 2. The lineal relationship confirmed the Microplate Spectrophotometer method was applicable to measure n-butanol concentration.

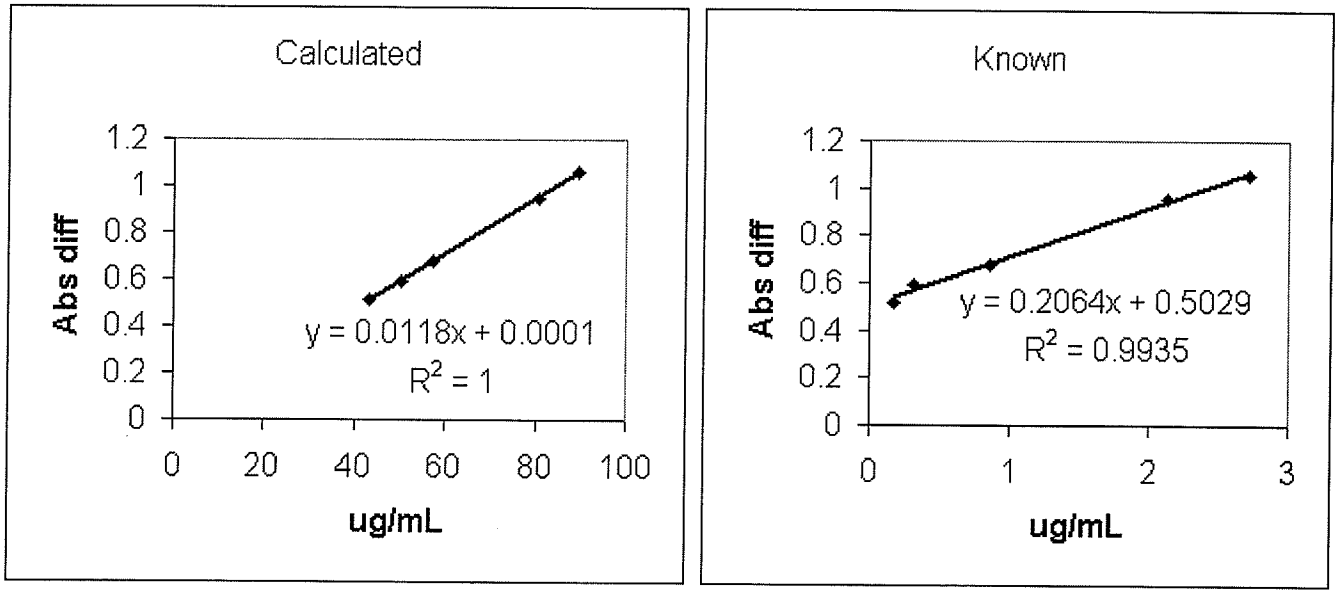


Figure 3. Lineal relationships between n-butanol concentration and absorbance difference for both calculated concentration and know concentration.

5. RESULTS AND DISCUSSION

5.1 Aerodynamics in the wind tunnel

5.1.1 Velocity profile

The measured velocity profiles exhibited a substantial degree of non-uniformity inside the hood as shown in figs. 4 to 11 and Table 3. Take bulk velocity of 0.3 m/s for instance, in the vertical direction, the velocity was higher at the top and bottom than in the middle (fig. 4). The largest difference occurred at X = 400 mm, where the velocity near the emission surface (0.36 m/s) was 33% higher than that in the mid height of the hood (0.27 m/s) and the coefficient of variation (CV) was 14%. On the horizontal plane, the velocity along the sides of the hood was much higher that that in the center (fig. 5). At X = 400 mm, the velocity along the left-hand side was 0.74 m/s which was 9.5 times that in the center (0.08 m/s) and the CV was 82%. However, the velocities at the center of the wind tunnel were relatively uniform. The similar non-uniformity of velocity was observed at bulk velocities of 0.20 (figs. 6 and 7), 0.24 (figs. 8 and 9) and 0.38 m/s (figs. 10 and 11).

Table 3: Coefficients of variation (CV) of measured velocities on vertical and horizontal planes at various locations in the wind tunnel for bulk velocities (BV) of 0.20, 0.24, 0.30 and 0.38 m/s

	Vertical Direction			Horizontal Direction		
Location	200 mm	400 mm	600 mm	200 mm	400 mm	600 mm
BV = 0.20 m/s	10%	12%	7%	44%	31%	37%
BV = 0.24 m/s	14%	15%	14%	50%	48%	46%

BV = 0.30 m/s	7%	14%	7%	66%	82%	74%
BV = 0.34 m/s	16%	17%	17%	82%	101%	94%

Comparing the velocity distributions at 0.20 m/s with those at 0.38 m/s, it was observed that the lower the bulk velocity, the less the velocity variation. In the vertical direction, the coefficient of variation (CV) of measured velocities was 12% at bulk velocity of 0.20 m/s, compared to 17% at 0.38 m/s. On the horizontal plane, the coefficient variation was 31% at 0.20 m/s versus 101% at 0.38 m/s.

It was also noted that the air flow was slightly being developed as the distance (X) extends from 200 to 600 mm. Figs 2 and 3 at bulk velocity of 0.3 m/s were good examples at vertical and horizontal planes, respectively. Vertically, the coefficient of variation (CV) was small, 7% with standard deviation of 0.03 at X=200 mm compared to 7% with standard deviation of 0.02 at X=600mm. Similarly, the coefficient of variation (CV) was high, 66% with the standard deviation of 0.22 at X=200 mm versus 74% with standard deviation of 0.22 at X=600 mm. Considering the measurement uncertainty of hot wire anemometer, the above velocity differences were negligible. Similar trends of minor air flow development were observed at figs 6 to 11 for different bulk velocities as well.

The non-uniform air flow inside the hood warrant new improvement for the wind tunnel. Flat vanes were installed as one modification to address horizontal flow turbulence in UNSW tunnel (Jiang et al., 1995). Similar assumptions about effects of flat vanes on UNSW tunnel could not be made to the wind tunnel used at University of

Manitoba, mainly because the dimensions and operation of UM tunnel are different from the UNSW wind tunnel.

The substantial degree of velocity non-uniformity inside the hood implies the difficulty in obtaining representative samples to calculate real emission rates. The velocity has a significant effect on the boundary layer for mass transfer, which could be expressed by using the two-film theory as follows (Tchobanoglous et al., 2003).

$$r = K_L(C_s - C_L) \tag{7}$$

where:

r = rate of mass transferred per unit area per unit time

K_L = overall mass transfer coefficient

C_S = concentration of n-butanol at the interface in equilibrium with the partial pressure of n-butanol in the bulk gas phase

C_L = concentration of n-butanol in the bulk liquid phase

The overall mass transfer coefficient is influenced by the bulk velocity. As the velocity increases, the boundary layer becomes thinner and the transfer coefficient becomes bigger, causing more mass transfer from the solution to the air. Due to the non-uniform bulk velocities, transferred n-butanol in the gaseous phase is not completely mixed, which poses the difficulty collecting representative samples to calculate real emission rate.

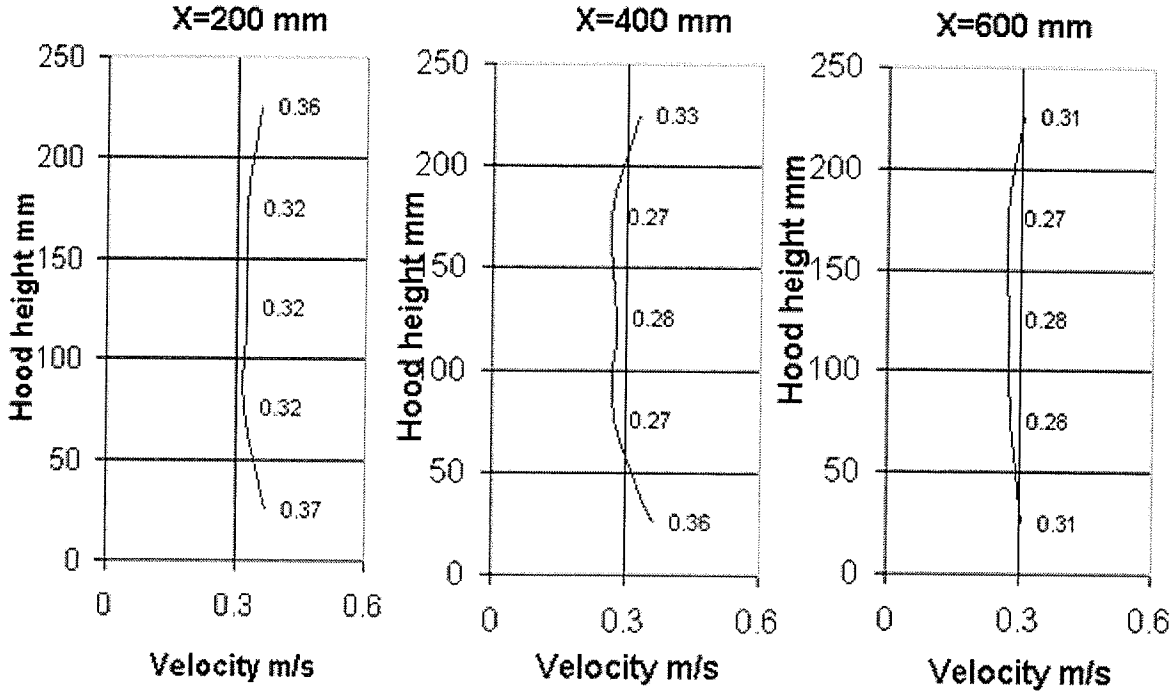


Figure 4. The velocity profiles in the vertical direction at bulk velocity of 0.3 m/s.

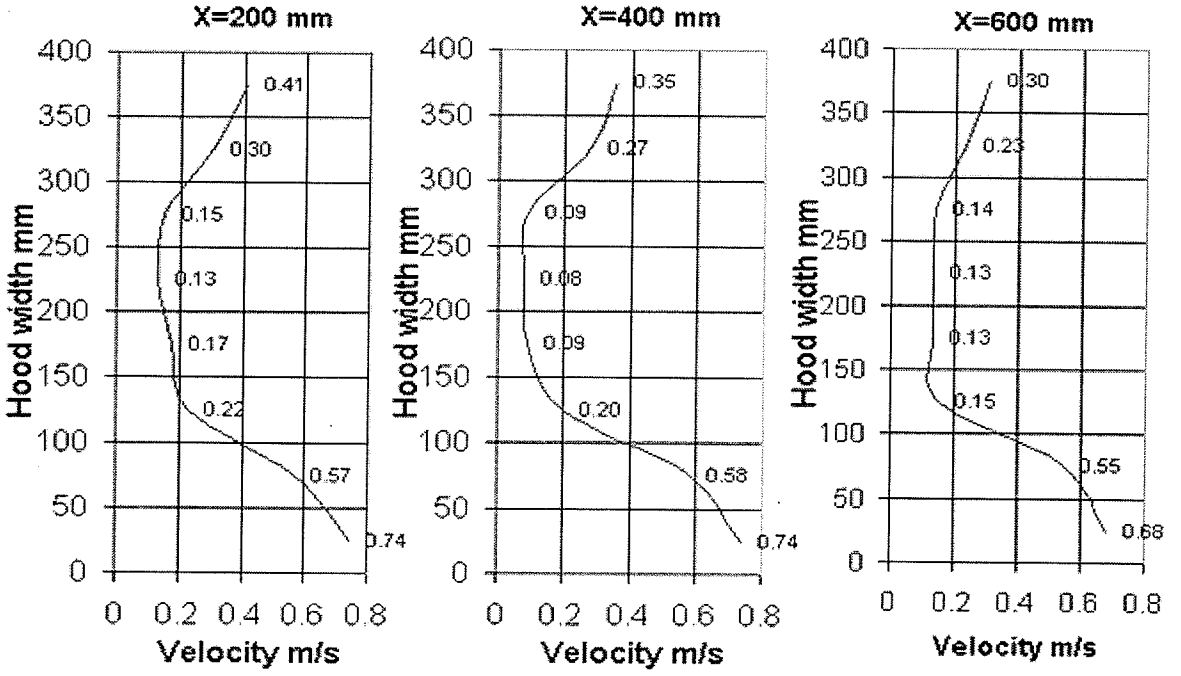


Figure 5. The velocity profiles in the horizontal direction at bulk velocity of 0.3 m/s.

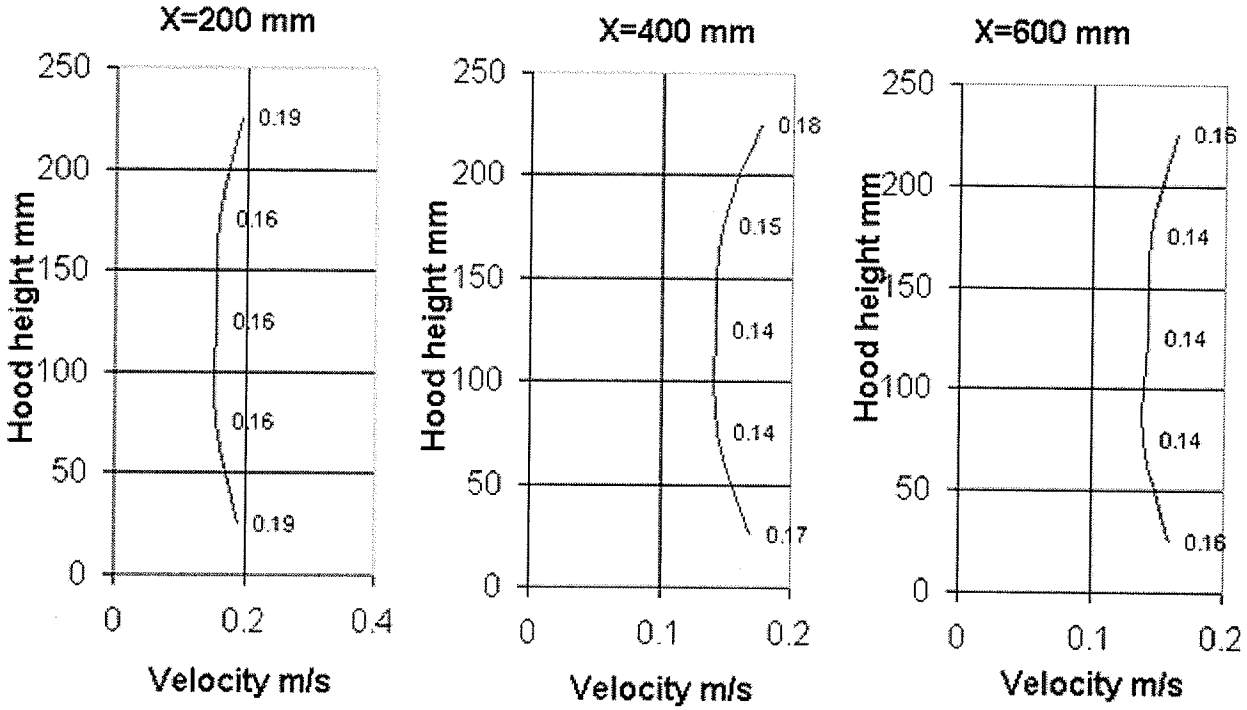


Figure 6. The velocity profiles in the vertical direction at bulk velocity of 0.2 m/s.

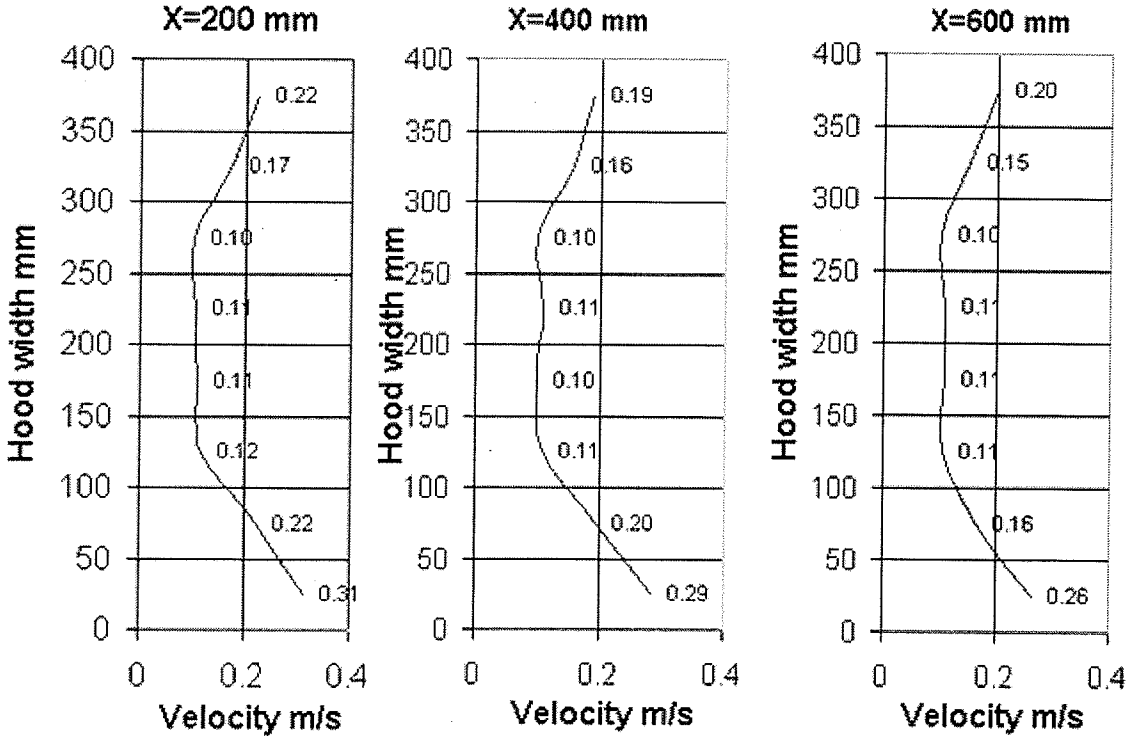


Figure 7. The velocity profiles in the horizontal direction at bulk velocity of 0.2 m/s.

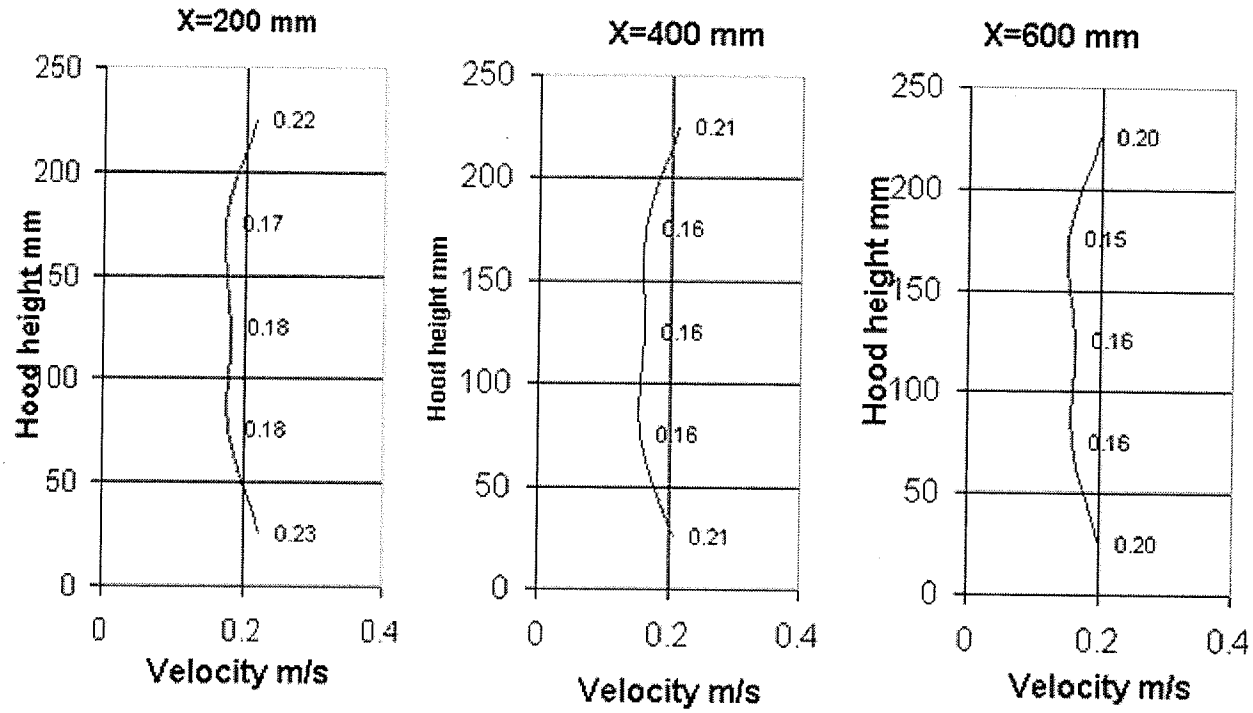


Figure 8. The velocity profiles in the vertical direction at bulk velocity of 0.24 m/s.

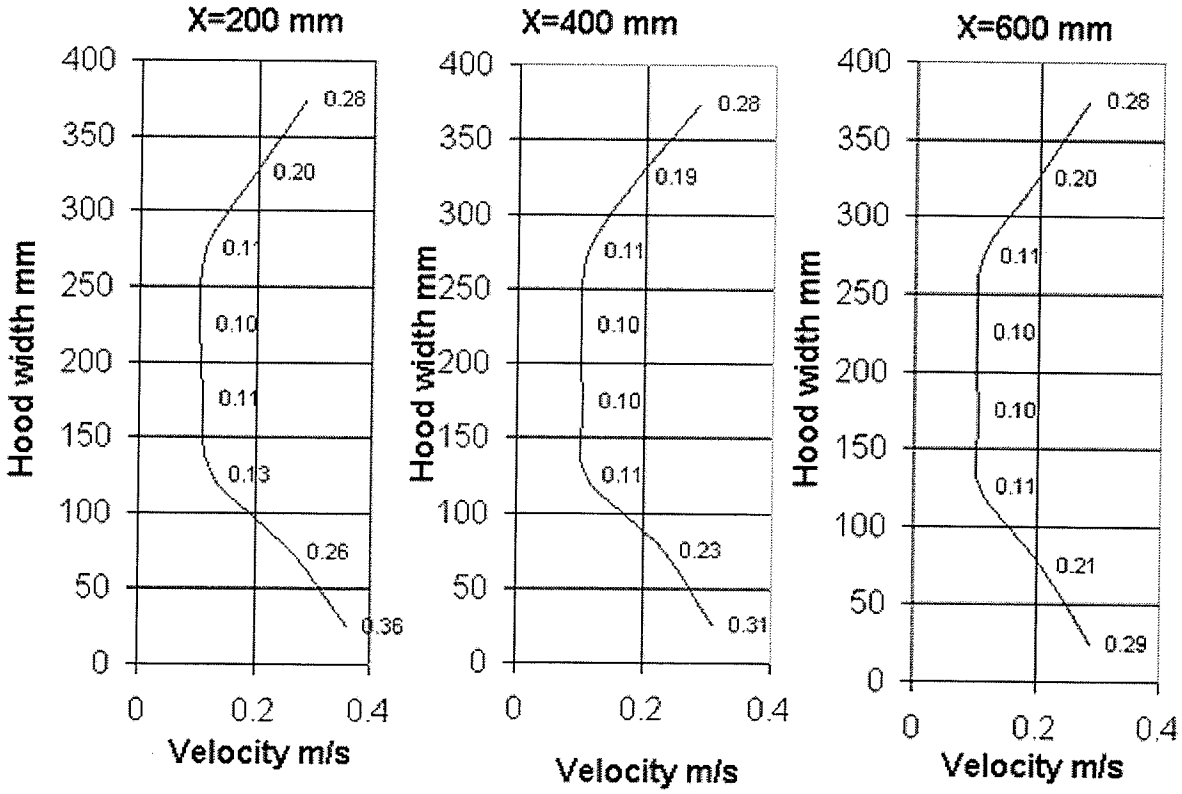


Figure 9. The velocity profiles in the horizontal direction at bulk velocity of 0.24 m/s.

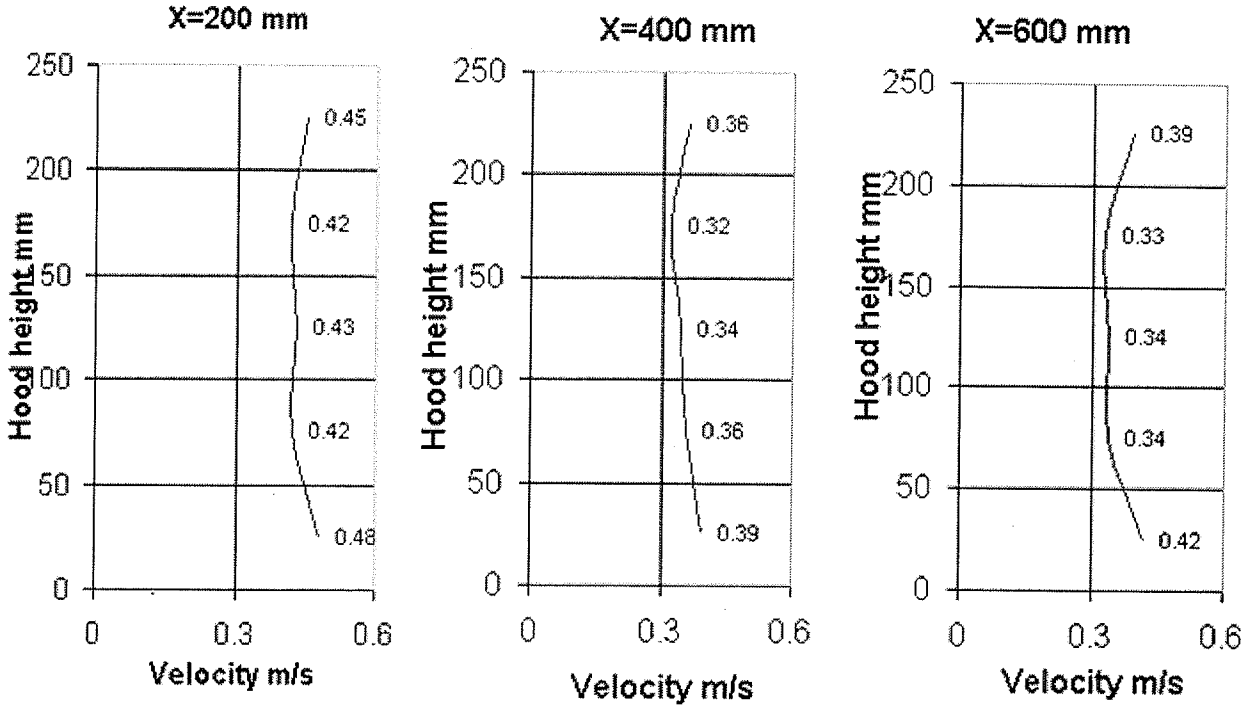


Figure 10. The velocity profiles in the vertical direction at bulk velocity of 0.38 m/s.

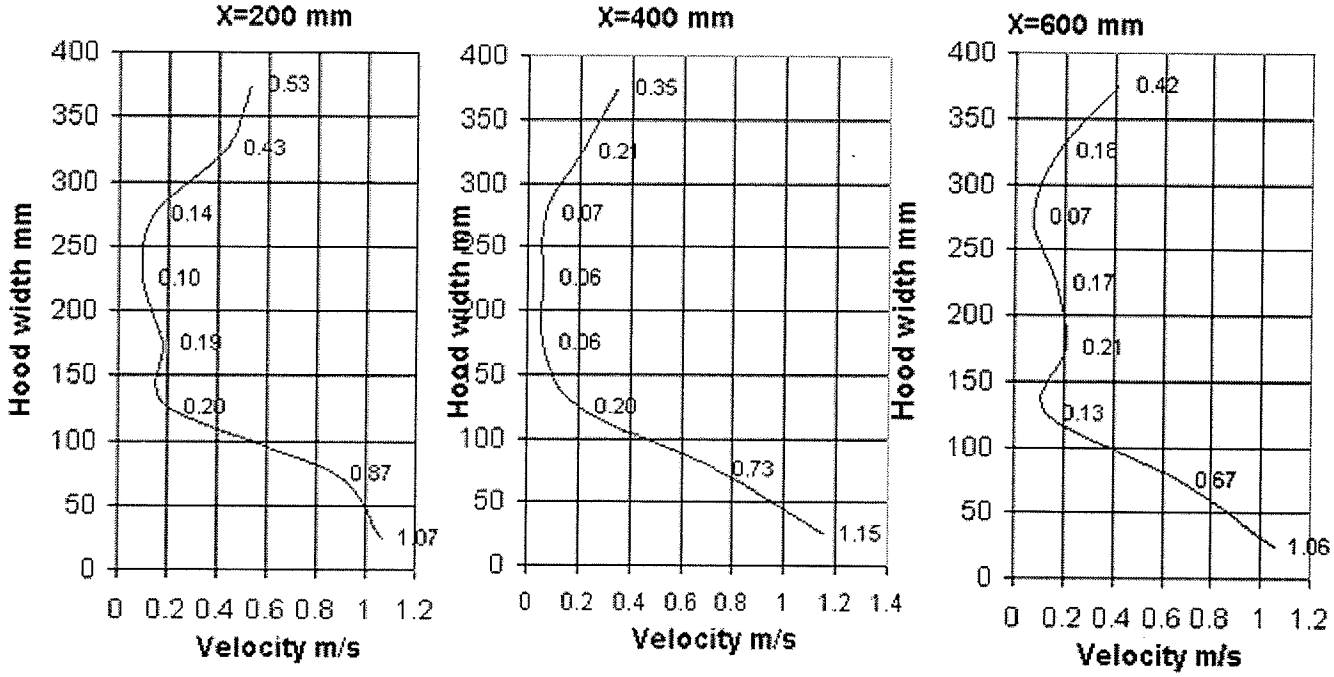
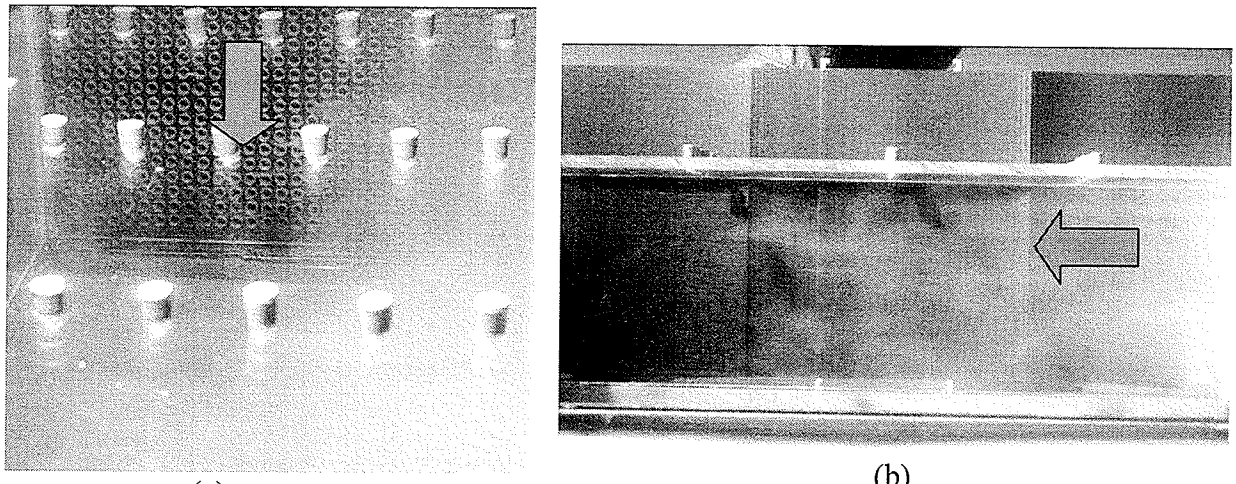


Figure 11. The velocity profiles in the horizontal direction at bulk velocity of 0.38 m/s.

5.1.2 Smoke test

The velocity profiles measured with the anemometer were qualitatively confirmed by the smoke tests. It was observed that the smoke came out first from the both sides of the baffle, and the smoke from the left side moved faster than the other side (fig. 12a). The front view indicated that the smoke moved faster at the top and bottom of the hood than the center (fig. 12b). The observations indicate that the velocity inside the hood was far from uniform, which were in accordance with the velocity non-uniformity conclusion from the velocity profile analysis.

The smoke testing also proved that the air flow inside the hood was turbulent. In fig. 12a, the smoke at both sides had hit the end of the tunnel before the smoke came out from the centre, indicative of significant velocity difference.



(a) (b)

Figure 12. Smoke flow patterns. (a) top view, (b) front view.

5.1.3 Reynolds Number calculation

The calculated Reynolds Number also showed turbulences in the wind tunnel hood. Using the lowest bulk velocity of 0.20 m/s, the Reynolds Number was calculated to be 4850 by using equation 7 (William, 1993), which is higher than the laminar flow limit of 2100.

$$R_e = \frac{\rho v_s L}{\mu} \tag{7}$$

where:

R_e = Reynolds Number

v_s = mean fluid velocity

L = hydraulic diameter (2*0.4*0.25/(0.4+0.25) = 0.31 m)

μ = (absolute) dynamic fluid viscosity

ρ = fluid density.

5.1.4 CFD modeling

The CFD simulations showed that the velocity distribution at the centre was relatively uniform while the velocity along the top and bottom of the hood was higher. (figs. 13 and 14). The velocity at the centre was around 0.26 m/s compared to velocity range of 0.32 to 0.52 m/s along the hood top and bottom. Similar Simulation results were also obtained for bulk velocities of 0.20 (figs. 15 and 16), 0.24 (figs. 17 and 18) and 0.38 m/s (figs. 19 and 20). This was in close agreement with the measured velocity profiles shown from figures 4 to 11.

By comparing the patterns for bulk velocity of 0.2 m/s (figs. 13 and 14) and for bulk velocity of 0.38 m/s (figs. 17 and 18), it was confirmed that the lower the bulk velocity, the less the velocity variation.

However, the velocity variation between tunnel center and sides could not be clearly illustrated in the results of the CFD simulations in comparison with velocity profile measurements. One possible reason might be accuracy limit of CFD modeling because CFD is initially designed to simulate relatively high velocities (more than 1 m/s), like building wind tunnel. The variation may become more significant if the bulk velocities are increased by one order.

The application of CFD simulation on velocity distribution inside the hood exhibits potentials for further research. CFD modeling provides an easy, quick and cost-effective way to test different tunnel development methods. The effect of methods on velocity distribution could be simulated before implementing the practical developments. It is recommended that CFD modeling is employed in further research to improve the velocity uniformity for UM wind tunnel.

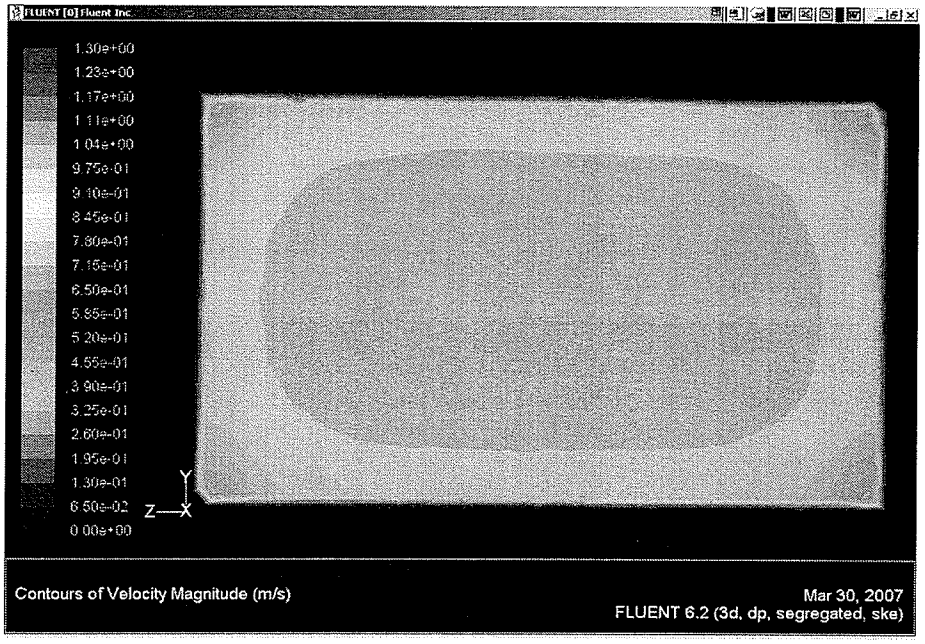


Figure 13. CFD simulations of velocity profile at bulk velocity of 0.30 m/s (front view)

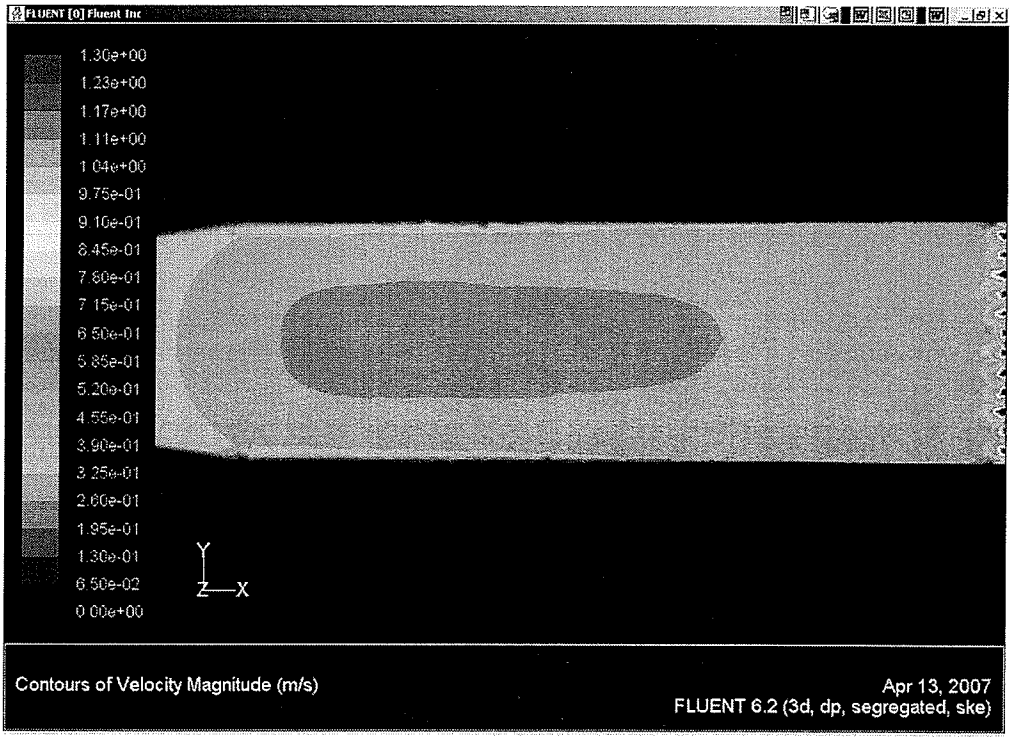


Figure 14. CFD simulations of velocity profile at bulk velocity of 0.30 m/s (top view)

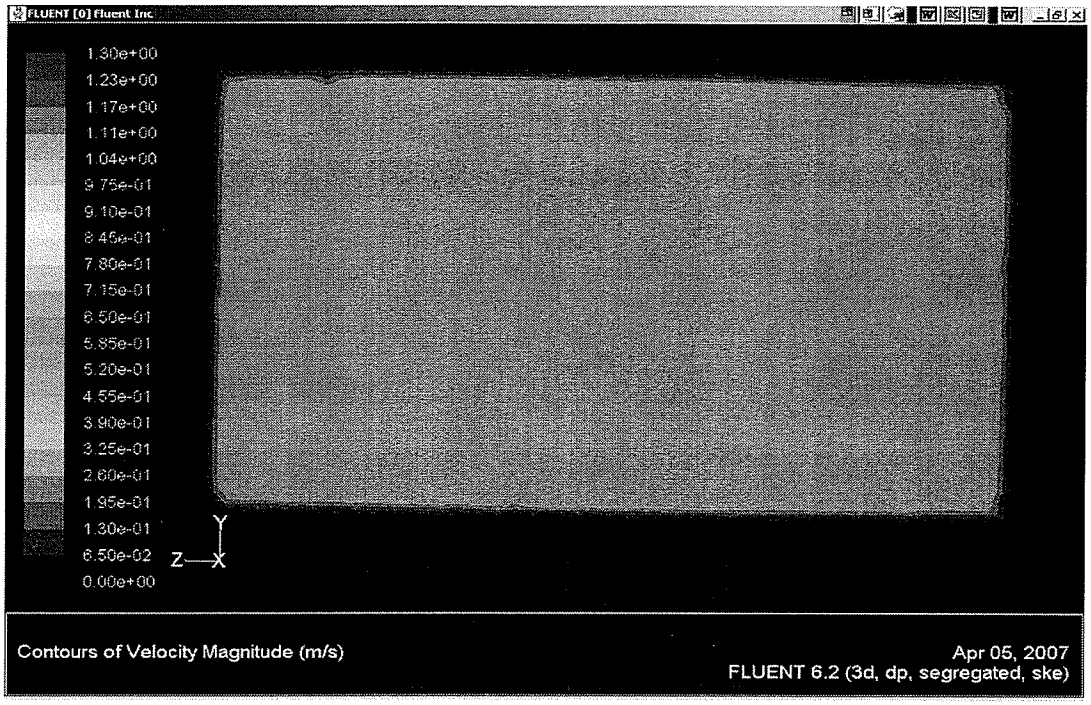


Figure 15. CFD simulations of velocity profile at bulk velocity of 0.20 m/s (front view)

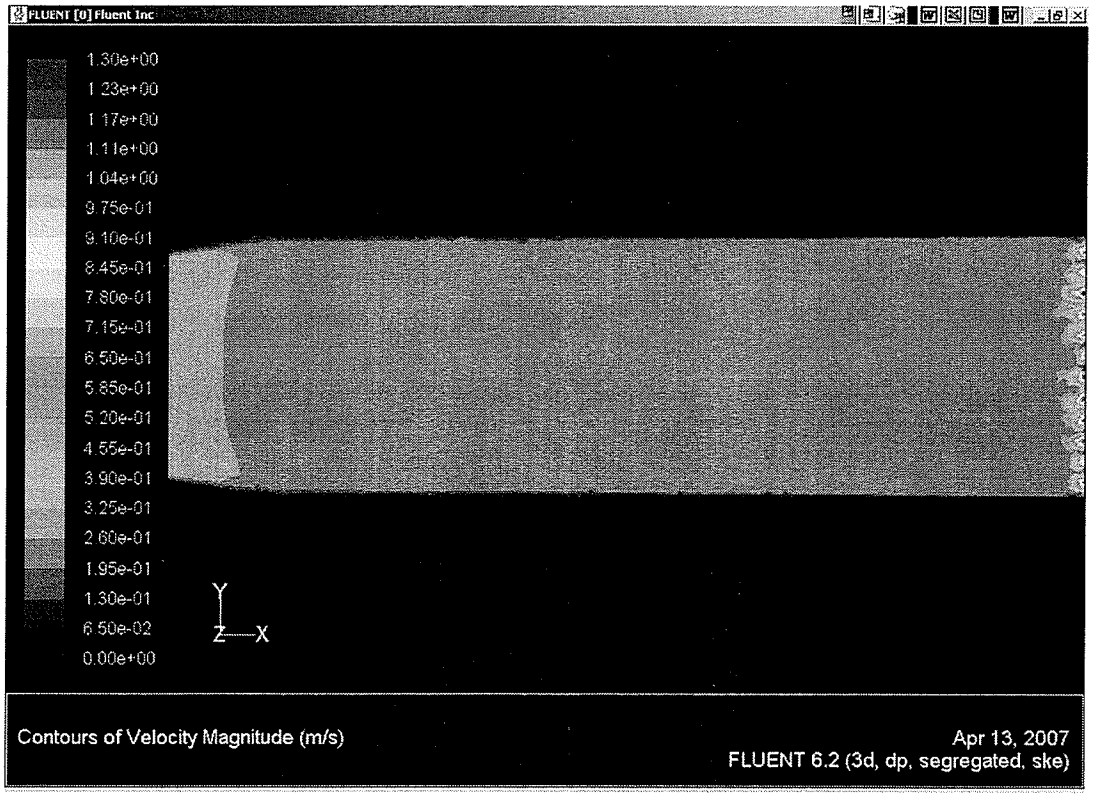


Figure 16. CFD simulations of velocity profile at bulk velocity of 0.20 m/s (top view)

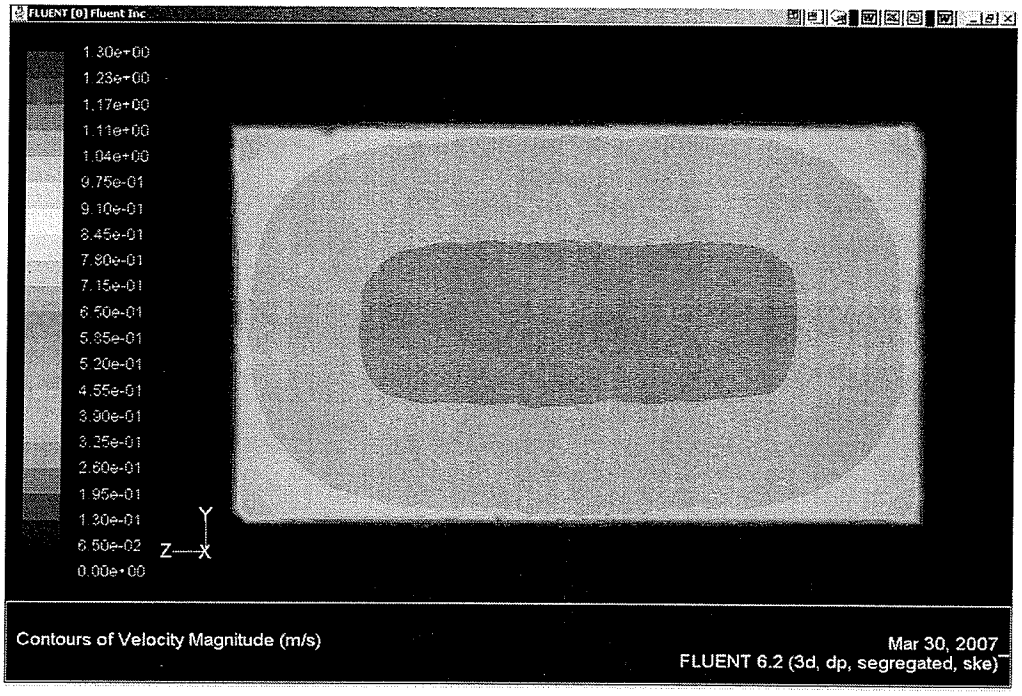


Figure 17. CFD simulations of velocity profile at bulk velocity of 0.24 m/s (front view)

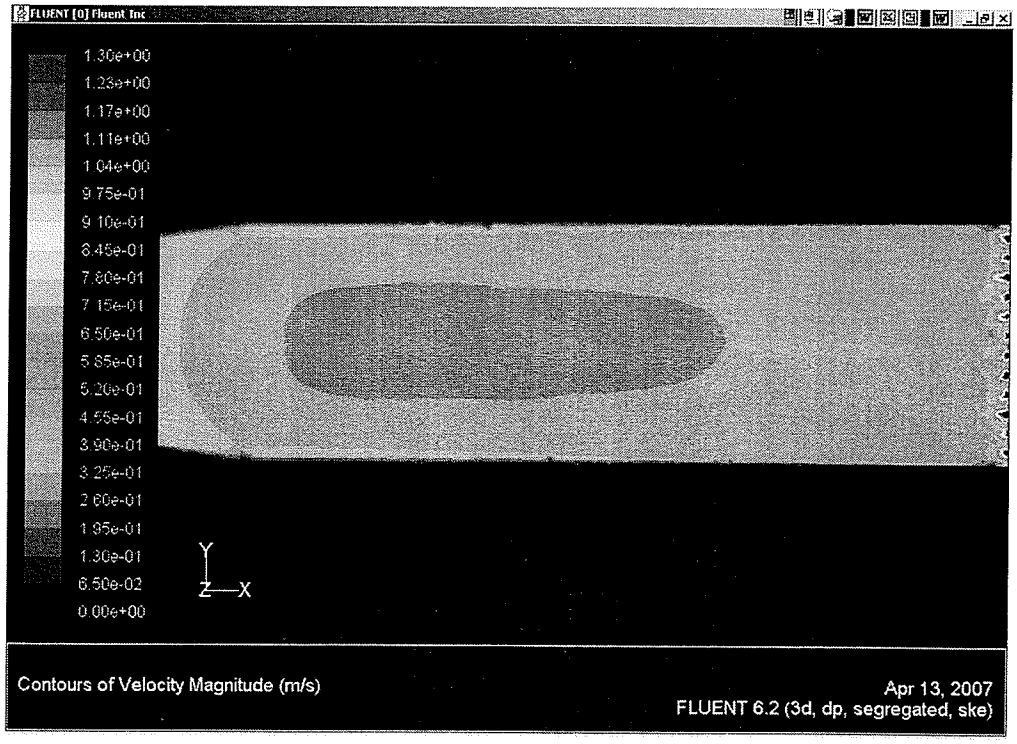


Figure 18. CFD simulations of velocity profile at bulk velocity of 0.24 m/s (top view)

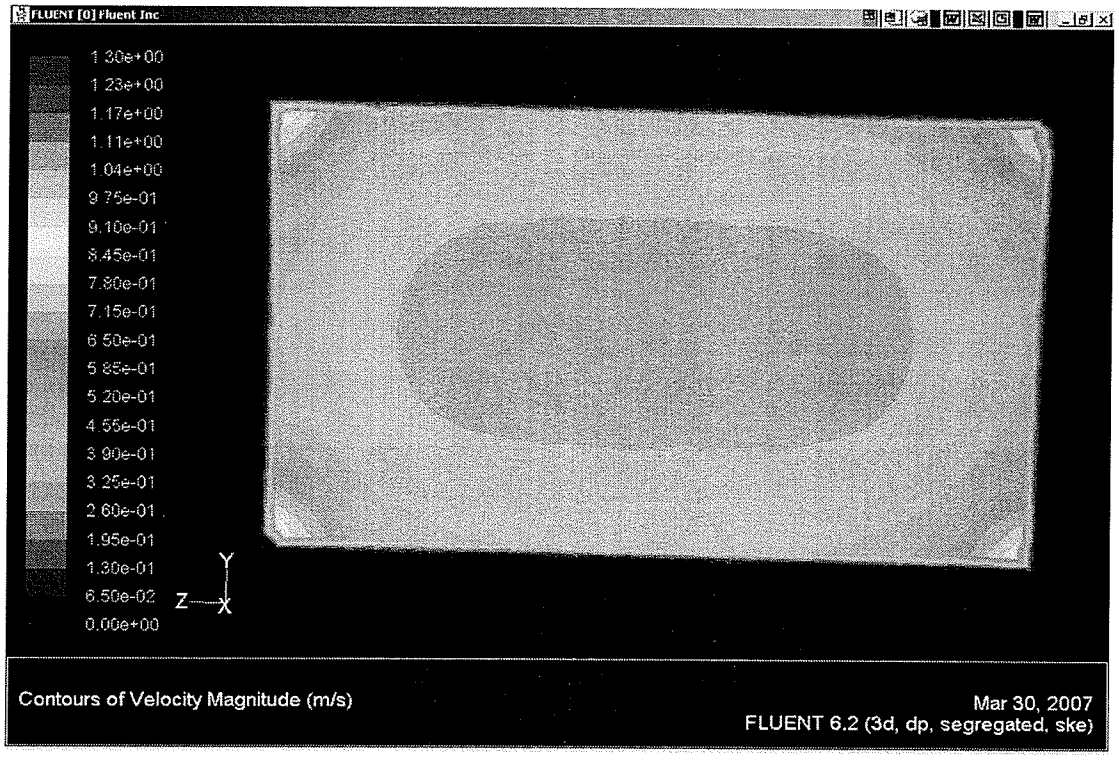


Figure 19. CFD simulations of velocity profile at bulk velocity of 0.38 m/s (front view)

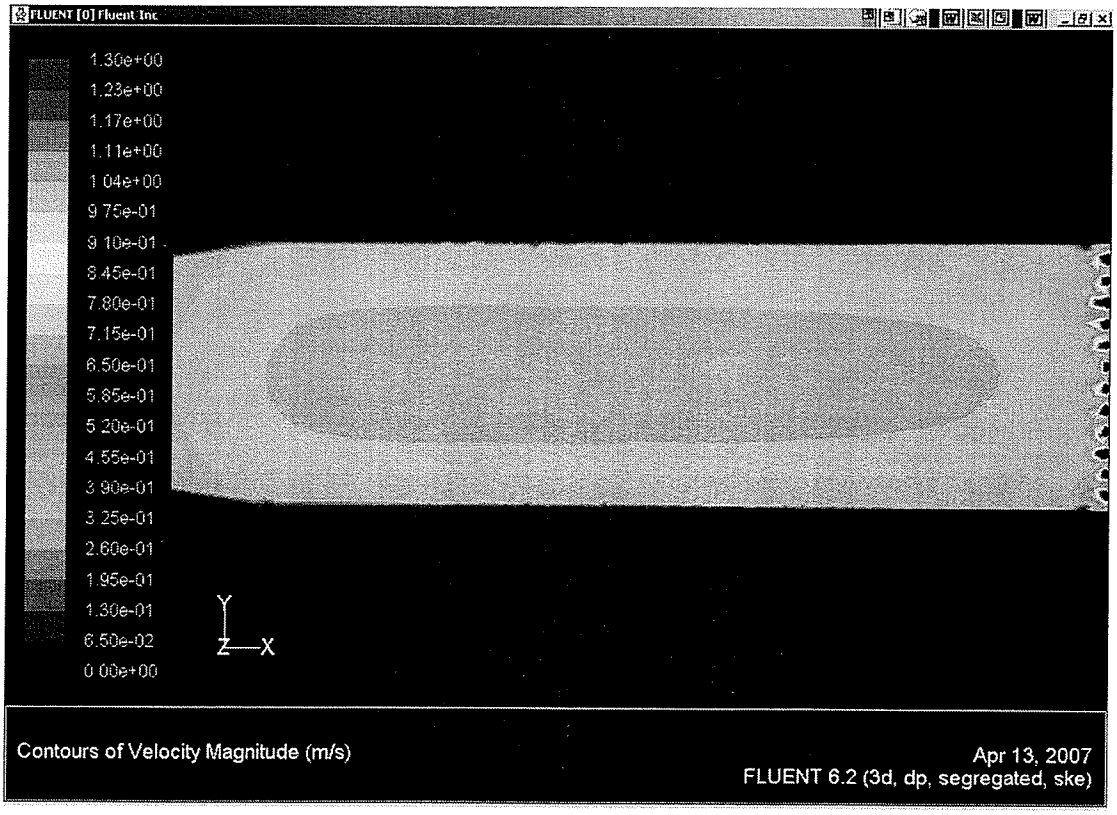


Figure 20. CFD simulations of velocity profile at bulk velocity of 0.38 m/s (top view)

5.2 Pick-up rate (Recovery efficiency)

Pick-up rate was calculated as the ratio of gaseous n-butanol amount captured at the sampling port to the liquid n-butanol loss from the tank at a 10-minute interval. Figure 21 shows the pick-up rates at different bulk velocities and n-butanol concentrations. The rates ranged from 37% to 56%. In other words, only 37% to 56% of n-butanol that transferred from the solution to air was captured by the gas samples taken at the exit of the wind tunnel. The low pick-up rate was possibly attributable to using one sampling point at the wind tunnel exit. The distribution of n-butanol might be highly non-uniform at the exit. Taking gas samples at on point (mid-stream) would not represent the average concentration of n-butanol in the exit gas. Multi-point sampling should be used to improve the pick-up rate. Loubet et al. (1999) reported that the "simulated" recovery efficiency of a 1-point, a 20-point sampling port with equally-spaced distribution, and 20-point sampling port with quadratic distribution were 61%, 89% and 100.4%, respectively. Sohn et al. (2005) conducted a similar experiment to determine the effect of sampling port design on the gas recovery efficiency. The results exhibited that the sampling recovery efficiency using one-point sampling port ranged from 20 to 81%, averaged 49%, in contrast to the recovery efficiencies of 64% to 90% (average 71%) using multi-point sampling. Wang et al. (2001) conducted a similar experiment to study recovery efficiencies of emission for UNSW wind-tunnel systems. They reported that the initial recovery efficiency from a simulated surface area emission source was only 37% to 48% by mass balance using carbon monoxide as the tracer gas and the poor recoveries resulted from uneven mixing at the sample collection point. Wang et al., (2001) therefore developed a special sampling chamber extension and a sampling manifold with optimally

distributed sampling orifices for the wind-tunnel sampling system and the recovery efficiency of sampling was improved to 83-100% with an average of 90%,. When the CO tracer gas was introduced at a single point, the recovery efficiency ranged from 92% to 102%, with an average of 97%.

Another reason for the low pick-up rates was the possible variations in n-butanol concentration in the tank. The n-butanol concentrations measured near the surface was used in equation 5 for calculating the pick-up rate, assuming the concentration was uniform in the tank. If a concentration gradient existed (i.e., lower at the surface) after running the wind tunnel for 10 minutes, using the concentration measured near the surface would overestimate the amount of n-butanol loss from the solution in the tank, which in turn would result in a lower pick-up rate.

According to Figure 21, in general, pick-up rate decreased as the velocity and solution concentration increased. The strongest simulated odour source (3488 ppm) at highest bulk velocity exhibited smallest pick-up rate. The variance analysis showed that velocity had a significant effect on pick up rate ($P<0.05$) while concentration did not affect the pick-up rate significantly ($P>0.05$).

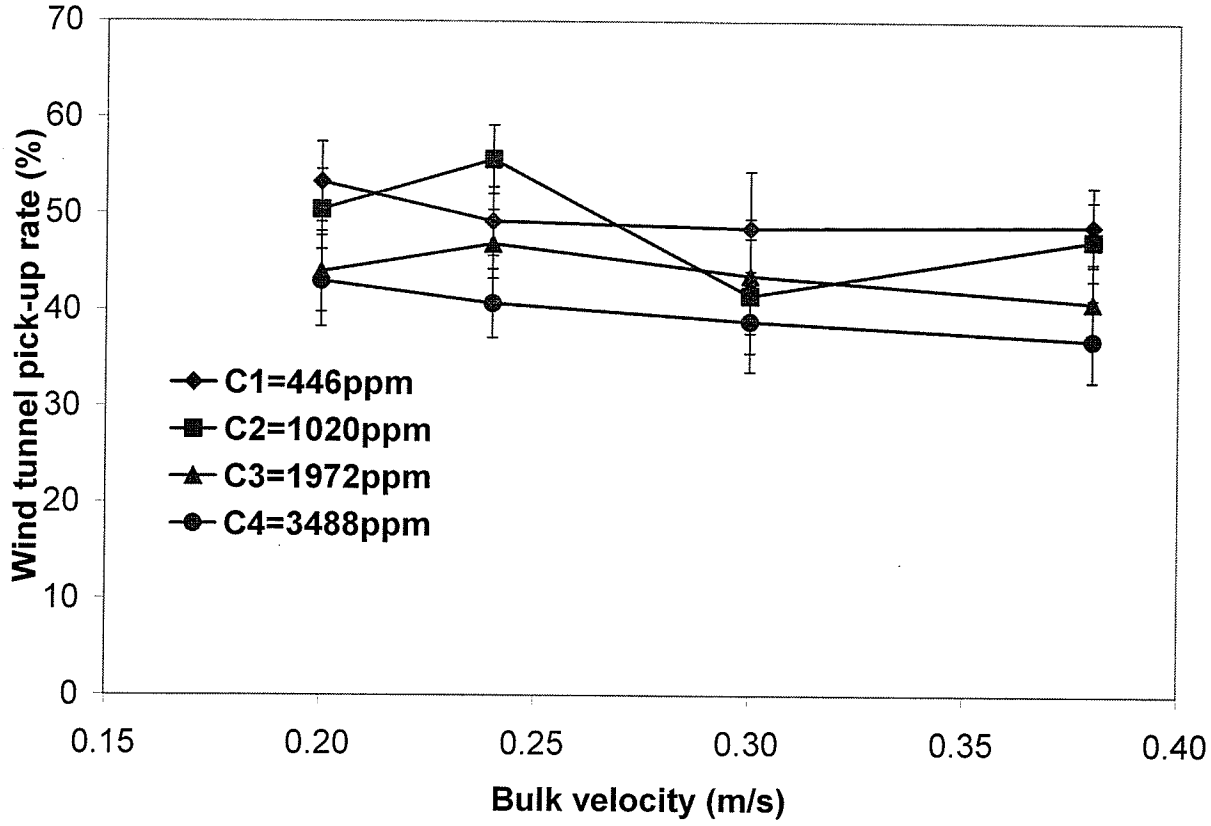


Figure 21. Pick-up rates at bulk velocities of 0.20, 0.24, 0.30, 0.38 m/s and different n-butanol solution concentrations (the error bars indicate standard deviations).

5.3 Odour emission rate

In the European odour standard, odour concentration (OU_E/m^3) is defined as equivalent to 123 μg n-butanol evaporated into 1 cubic meter of neutral gas at standard conditions (CEN, 2003). Thus, n-butanol concentration measured (ppm) may be converted to odour concentration at 1 atmospheric pressure and 25°C based on the n-butanol release from the solution through equation 8:

$$E = \frac{M_n}{T \times A_s \times 123} \tag{8}$$

where:

E = odour emission rate ($OU_E/s\text{-}m^2$)

M_n = n-butanol mass picked up by air flowing through the wind tunnel (μg)

T = time interval (s)

A_s = surface area covered by the tunnel (0.32 m^2)

Figure 22 shows the relationship between the odour emission rate and the bulk velocity in the hood derived from the n-butanol release from the solution. It can be seen that the odour emission rate increased with the bulk velocity. This was because the boundary layer above the solution surface became thinner, thus releasing more n-butanol from the solution to the air as the bulk velocity increased. The variance analysis showed both the bulk velocity and solution concentration had significant effect on the odour emission rate ($P<0.05$). The simulated odour source with the largest concentration (3488 ppm) had the highest emission rate increase for all velocities tested. It was because more n-butanol transferred from the solution to air at a higher solution concentration.

The relationship between odour emission rate and bulk velocity in the hood could be described by the power function. The exponent varied from 0.72 from 0.77 when the solution concentration changed from 446 to 3488 ppm. These values are comparable to those reported by Smith and Watts (0.63) (1994b) and Schmidt (1999) (0.89).

The odour emission rate based on the gaseous n-butanol measurement at the exit could be calculated using equation 9 and 10.

$$C_E = \frac{M}{24.5} \times C_{ppm} \times \frac{1000}{123} \tag{9}$$

where:

C_E = odour concentration (OU_E/m^3)

M = n-butanol molecular weight (74.12g/mol)

C_{ppm} = n-butanol concentration expressed at ppm

According to Equation 9, odour emission rates at different wind speed were calculated using equation 10.

$$E = C_E V_t \frac{A_t}{A_s} \tag{10}$$

where:

C_E = odour concentration (OU_E/m^3)

V_t = the bulk wind speed in the tunnel

A_t = the cross-sectional area of the tunnel ($0.1m^2$)

A_s = the surface area covered by the tunnel ($0.32m^2$)

Figure 23 shows the relationship between the odour emission rate and the bulk velocity in the hood derived from the gaseous n-butanol concentration measurement at

the exit. It shows that the emission rate increased with the bulk velocity and both velocity and solution concentration had significant effect on the emission rate ($P<0.05$).

It is interesting to note that while both the emission rates determined from the gaseous n-butanol concentration at the exit and from the n-butanol concentration of solution increased with the bulk velocity, the pick-up rate, determined as the ratio of the two emission rates, decreased with the bulk velocity (fig. 21). As the bulk velocity increased, the air flow inside the tunnel became more turbulent and better mixed at the exit. This would reduce the sampling error caused by the use of a single port at the exit. If the single port sampling was the dominant cause of low pick-up rates, the pick-up rate would increase with the bulk velocity. The observed decrease in the pick-up rate with the bulk velocity seemed to indicate that the overestimate of n-butanol released to air, which was due to the concentration gradient in the solution, might be the dominant cause of low measured pick-up rates.

There are several other factors which may affect the emission rate besides the bulk velocity. Other factors include differences in the material used in constructing the tunnel, the length to width ratio, the surface area sampled and the height of hood. Smith and Watts (1994b) compared odour emissions rates from a feedlot using a small (1m long by 0.25 m wide and 0.2 m high) and large (2 m long by 0.5 m wide and 0.45 m high) wind tunnel. The emission rates measured with the large wind tunnel were consistently lower than those in the smaller tunnel by a factor of 0.8. It was suggested that this difference was largely due to the difference in velocity profiles of the tunnels, e.g., a steeper velocity profile gradient in the smaller tunnel resulting in a higher wind speed near the surface.

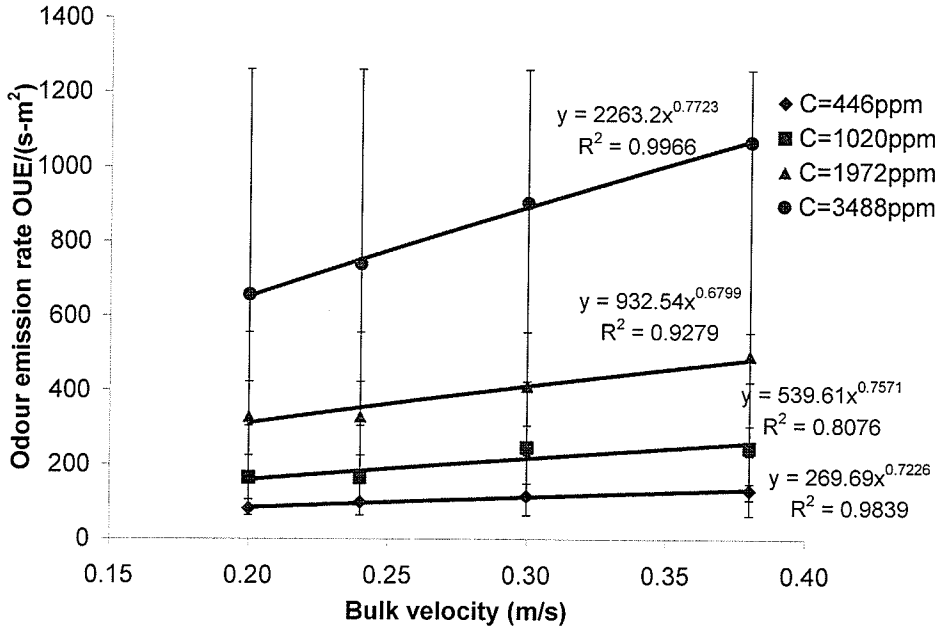


Figure 22. Relationship between odour emission rate and bulk velocities derived from n-butanol release from the solution (the error bars indicate standard deviations).

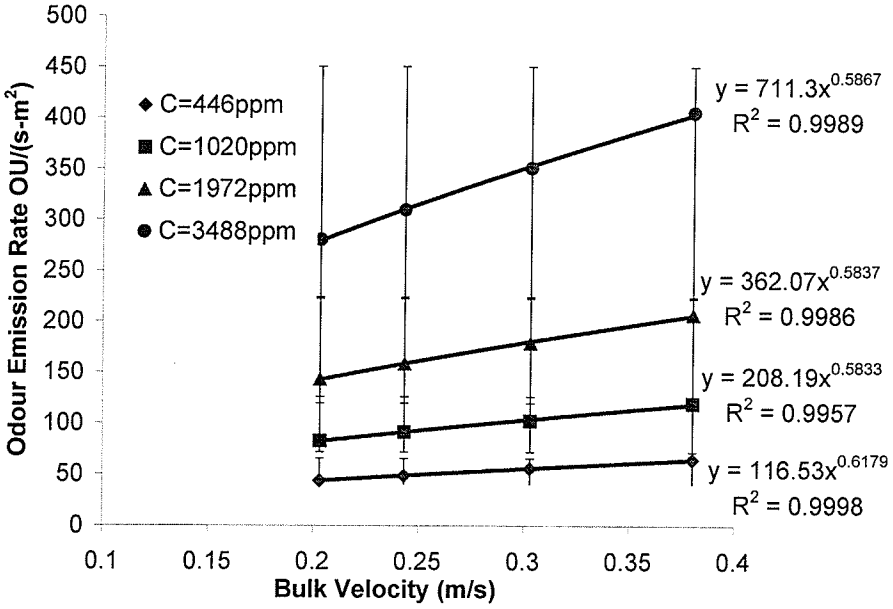


Figure 23. Relationship between odour emission rate and bulk velocities derived from the gaseous n-butanol measurement at the exit (the error bar indicate standard deviations).

6. CONCLUSIONS

A wind tunnel for sampling odour from liquid emission surfaces was evaluated under the controlled laboratory conditions. The evaluation process consisted of air velocity measurement inside the hood, smoke testing, CFD simulations of air movement in the hood, and pick-up rate (recovery efficiency) estimation. The following conclusions could be drawn from this study:

1. The measured velocity profiles exhibited a substantial degree of non-uniformity. The air tended to move faster along the sides of the hood than the center. The lower the bulk velocity, the less the velocity variation
2. The air flow inside the hood of the wind tunnel was turbulent. Reynolds Number was calculated to be 4850 at the lowest bulk velocity of 0.20 m/s in the experiment, which is greater than the limit of 2100 for laminar flow.
3. The current sampling system measured 37% to 57% of n-butanol released from the solution to the tunnel. There exist many factors affecting the pick-up rate. One possible reason for the low pick up rate was the use of a single sampling port at the exit for measuring the n-butanol concentration in the air because the n-butanol concentration at the exit of the wind tunnel hood might not be uniform.
4. The odour emission rate measured by the wind tunnel increased with the bulk air velocity in the tunnel and the relationship could be described by the power function.

7. RECOMMENDATION FOR FUTURE RESEARCH

The laboratory evaluation of the wind tunnel used at University of Manitoba revealed several deficiencies in the current design. These deficiencies include non-uniform air velocity inside the hood and poor odour pick-up rates. Further research, therefore, is recommended as follows:

1. Further developments on the wind tunnel are needed to improve the dynamic performance and the recover efficiency. The use of flow enhancing devices, such as flat vanes suggested by Jiang et al. (1995) (fig. 24), should be explored to improve the air flow patterns in the wind tunnel. Multi-point sampling systems have been confirmed to increase the recovery rate.
2. The average pick-up rate for the wind tunnel was only 37% to 56%. One reason for the low pick-up rate may be using only one point sampling of gas concentration at the exit, which could not collect representative average concentration. Multi-point sampling systems should be considered for wind tunnel sampling. Another reason of low pock-up rate might be attributable to the possible concentration gradient in the solution during 10 minute experiment interval. If a gradient existed, the n-butanol released from the solution would be overestimated, thus resulting in a low pick-up rate. In the future research, the concentration profile in the n-butanol solution should be monitored during the entire period of each experiment.

3. An idea wind tunnel should have uniform velocity profiles within the tunnel and 100% pick-up (recovery) rate. I recommend wind tunnel design that will have flow enhancing devices, such as flat vanes and baffles, a longer inlet pipe, and a multi-point sampling system with quadratic distribution (fig. 24). A fan could also be added at the exit to promote mixing.

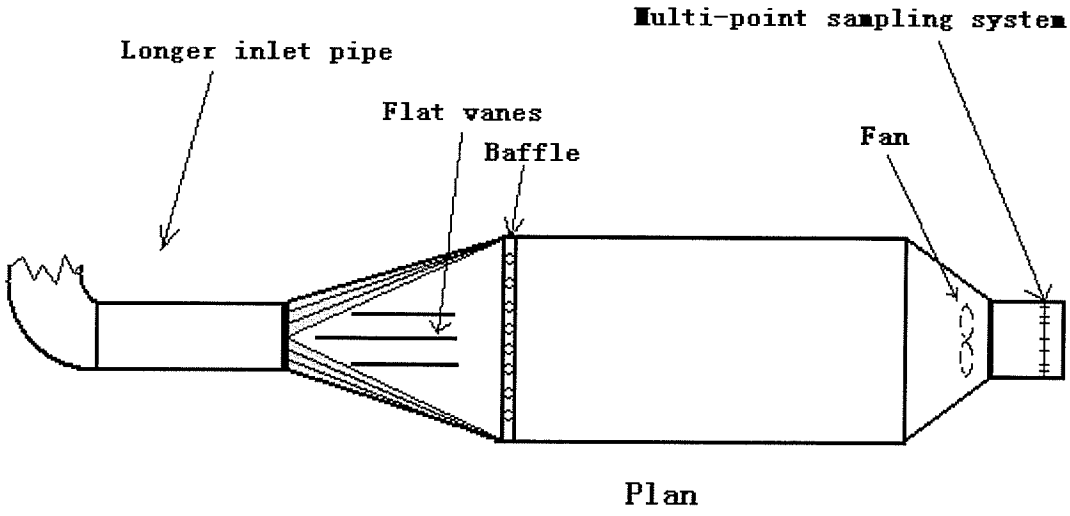


Figure 24. Schematic drawing of an idea wind tunnel.

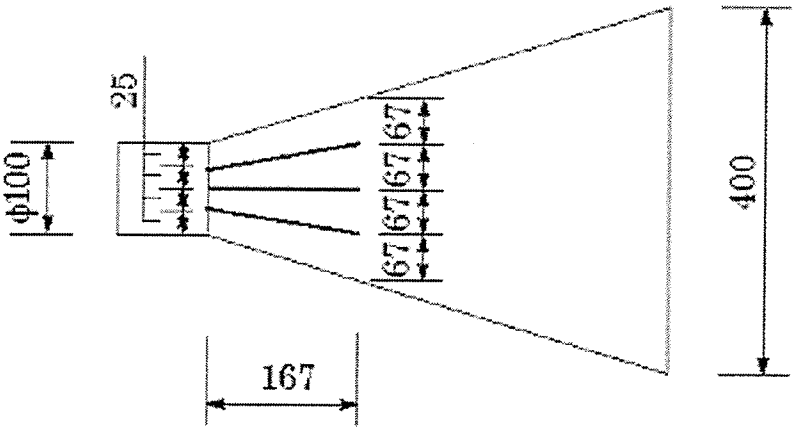


Figure 25. Schematic drawing of flat vanes installation (all dimensions are in mm) (Courtesy of Jiang et al., 1995).

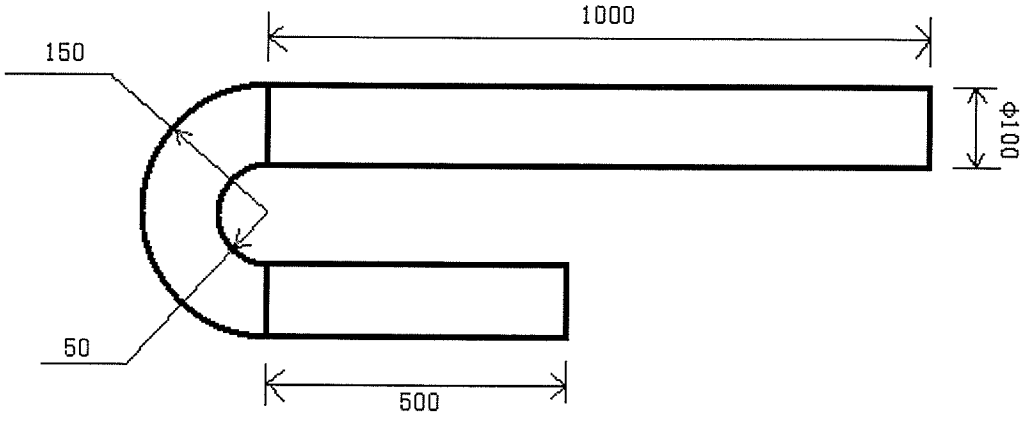


Figure 26. Schematic drawing of longer inlet pipe (all dimensions are in mm).

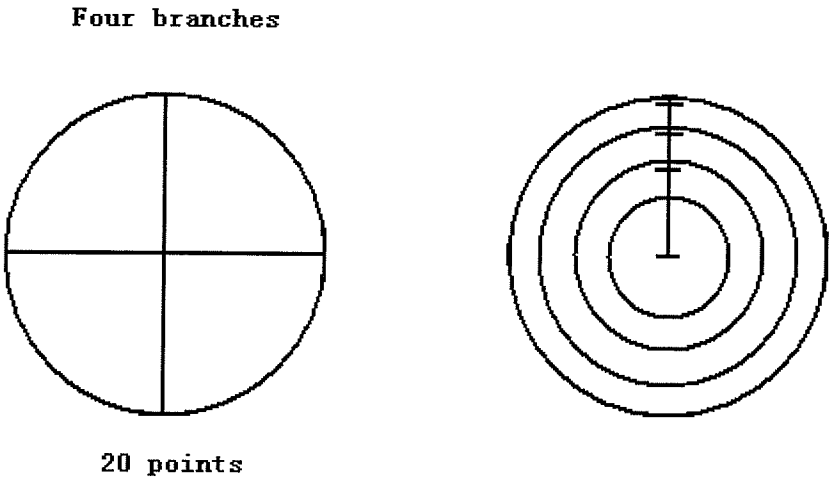


Figure 27. Schematic representation of the quadratic distribution of samples points along a branch and its related area (Courtesy of Loubet et al., 1999a).

4. Computational Fluid Dynamics (CFD) modeling provides a cost-effective means to simulate air movement behavior in wind tunnels. It is recommended that systematical CFD simulations are conducted to study and compare the performance and behavior of various wind tunnel designs. The CFD should be used as a tool to assess the performance and behavior of new wind tunnel designs.

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APPENDICES

Appendix A
Velocity Measurement Data

Velocity: 3.84m/sec(duct); 0.3m/sec (surface)																								
Temperature: 22.1 – 24°C																								
No. / Depth	Group I								Group II								Group III							
	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8
25cm	0.72	0.55	0.24	0.27	0.15	0.28	0.29	0.36	0.78	0.59	0.28	0.18	0.15	0.1	0.32	0.23	0.51	0.61	0.18	0.18	0.19	0.18	0.13	0.48
75cm	0.68	0.47	0.3	0.17	0.16	0.12	0.27	0.43	0.71	0.51	0.11	0.03	0.04	0.07	0.29	0.37	0.64	0.46	0.1	0.16	0.18	0.1	0.19	0.34
125cm	0.77	0.58	0.23	0.12	0.06	0.11	0.33	0.37	0.74	0.52	0.13	0.03	0.03	0.07	0.31	0.39	0.67	0.53	0.1	0.11	0.13	0.1	0.27	0.29
175cm	0.68	0.57	0.15	0.13	0.14	0.08	0.34	0.46	0.71	0.58	0.17	0.04	0.04	0.07	0.22	0.35	0.77	0.47	0.15	0.13	0.09	0.17	0.27	0.18
225cm	0.87	0.67	0.2	0.18	0.15	0.18	0.29	0.42	0.77	0.71	0.29	0.18	0.13	0.15	0.23	0.43	0.81	0.66	0.22	0.07	0.06	0.13	0.29	0.23

Velocity: 2.52 m/s (duct); 0.2 m/s (surface)																								
Temperature: 22.3 – 23.6°C																								
No. / Depth	Group I								Group II								Group III							
	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8
25cm	0.29	0.35	0.16	0.05	0.09	0.08	0.25	0.26	0.31	0.24	0.13	0.05	0.07	0.09	0.28	0.24	0.21	0.26	0.06	0.09	0.11	0.1	0.18	0.3
75cm	0.22	0.19	0.11	0.15	0.12	0.18	0.14	0.17	0.27	0.15	0.06	0.12	0.12	0.14	0.14	0.17	0.27	0.15	0.12	0.16	0.09	0.11	0.1	0.15
125cm	0.28	0.15	0.12	0.13	0.14	0.15	0.13	0.15	0.26	0.11	0.1	0.12	0.12	0.14	0.13	0.16	0.22	0.16	0.13	0.1	0.13	0.11	0.15	0.14
175cm	0.24	0.22	0.1	0.11	0.11	0.07	0.12	0.27	0.23	0.13	0.16	0.12	0.12	0.11	0.12	0.16	0.21	0.12	0.13	0.12	0.11	0.09	0.17	0.17
225cm	0.53	0.21	0.1	0.1	0.07	0.02	0.22	0.27	0.36	0.35	0.1	0.09	0.11	0.02	0.12	0.21	0.41	0.13	0.09	0.07	0.1	0.09	0.15	0.24

Velocity: 3.03 m/sec (duct); 0.24m/sec (surface)																								
Temperature: 22.5 – 23.4°C																								
No. / Depth	Group I								Group II								Group III							
	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8
25cm	0.56	0.47	0.16	0.26	0.05	0.03	0.07	0.12	0.55	0.43	0.29	0.08	0.03	0.05	0.15	0.1	0.39	0.27	0.21	0.05	0.07	0.03	0.17	0.39
75cm	0.52	0.31	0.15	0.08	0.03	0.04	0.18	0.08	0.57	0.31	0.08	0.04	0.07	0.04	0.11	0.07	0.34	0.33	0.1	0.13	0.1	0.04	0.04	0.13
125cm	0.01	0.12	0.08	0.05	0.18	0.27	0.21	0.54	0.02	0.02	0.03	0.15	0.13	0.21	0.22	0.51	0.1	0.12	0.1	0.22	0.21	0.19	0.19	0.16
175cm	0.25	0.08	0.13	0.02	0.08	0.18	0.3	0.39	0.02	0.02	0.02	0.17	0.13	0.11	0.33	0.45	0.18	0.05	0.03	0.02	0.02	0.17	0.39	0.41
225cm	0.46	0.34	0.12	0.13	0.18	0.05	0.23	0.29	0.39	0.37	0.13	0.08	0.14	0.13	0.13	0.29	0.42	0.26	0.1	0.1	0.1	0.12	0.19	0.31

Velocity: 4.88 m/s (duct); 0.38m/s (surface)																								
Temperature: 22.2 – 23.6°C																								
No. / Depth	Group I								Group II								Group III							
	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8	1	2	3	4	5	6	7	8
25cm	0.93	0.89	0.25	0.14	0.15	0.17	0.5	0.57	0.98	0.76	0.36	0.06	0.06	0.08	0.21	0.37	1.08	0.67	0.12	0.19	0.15	0.18	0.25	0.5
75cm	0.96	0.82	0.17	0.16	0.08	0.14	0.43	0.58	1.12	0.68	0.12	0.03	0.08	0.07	0.18	0.3	0.98	0.56	0.12	0.22	0.16	0.05	0.13	0.42
125cm	1.08	0.78	0.17	0.15	0.05	0.17	0.47	0.58	1.13	0.71	0.13	0.03	0.07	0.06	0.3	0.3	1.08	0.59	0.1	0.2	0.15	0.03	0.12	0.44
175cm	1.18	0.83	0.13	0.13	0.08	0.12	0.37	0.52	1.23	0.76	0.13	0.05	0.06	0.06	0.25	0.3	0.89	0.67	0.12	0.24	0.16	0.06	0.15	0.41
225cm	1.18	1.03	0.27	0.35	0.13	0.12	0.37	0.39	1.3	0.76	0.26	0.13	0.05	0.06	0.11	0.47	1.26	0.87	0.19	0.18	0.23	0.03	0.24	0.34

Appendix B
Bio-teck data recording

Experiment 1: 250ppm solution concentration

Assay performed on March 20, 2007

	Initial	0min	10min	0min	10min	0min	10min	0min	10min	Blank	Control	
ABS(A1)	0.21	0.202	0.195	0.194	0.193	0.193	0.194	0.195	0.197	0.219	0.194	
	0.21	0.204	0.196	0.193	0.192	0.195	0.192	0.19	0.192	0.217	0.191	
	0.209	0.208	0.195	0.19	0.194	0.195	0.194	0.202	0.191	0.213	0.189	
ABS(A2)	0.533	0.532	0.517	0.512	0.511	0.512	0.507	0.507	0.501	0.251	0.78	
	0.542	0.53	0.518	0.515	0.509	0.511	0.504	0.506	0.499	0.245	0.78	
	0.538	0.529	0.518	0.516	0.513	0.513	0.508	0.51	0.5	0.249	0.81	
A2-A1	0.323	0.33	0.322	0.318	0.318	0.319	0.313	0.312	0.304	0.032	0.586	
	0.332	0.326	0.322	0.322	0.317	0.316	0.312	0.316	0.307	0.028	0.589	
	0.329	0.321	0.323	0.326	0.319	0.318	0.314	0.308	0.309	0.036	0.621	
Avg	0.328	0.325667	0.322333	0.322	0.318	0.317667	0.313	0.312	0.306667	0.032	0.598667	
C (g/L)	0.047026	0.048046	0.046881	0.046299	0.046299	0.046444	0.045571	0.045425	0.04426	0.004659	0.085317	
	0.048337	0.047463	0.046881	0.046881	0.046153	0.046007	0.045425	0.046007	0.044697	0.004077	0.085754	
	0.0479	0.046735	0.047026	0.047463	0.046444	0.046299	0.045716	0.044843	0.044988	0.005241	0.090413	
Cavg	0.047754	0.047415	0.046929	0.046881	0.046299	0.04625	0.045571	0.045425	0.044648	0.003106	0.058108	
C (mg/L)	47.75446	47.41474	46.92943	46.8809	46.29853	46.25	45.57056	45.42497	44.64848	3.105981	58.10773	
Original C(mg/L)	446.4848	443.0876	438.2345	437.7492	431.9255	431.4402	424.6458	423.1899	415.425			55.00175

Experiment 2: 500ppm solution concentration

Assay performed on March 23, 2007

	Initial	0min	10min	0min	10min	0min	10min	0min	10min	Blank	Control	
ABS(A1)	0.167	0.169	0.17	0.166	0.166	0.172	0.17	0.171	0.169	0.169	0.17	
	0.168	0.167	0.171	0.169	0.169	0.17	0.171	0.169	0.17	0.17	0.171	
	0.169	0.17	0.169	0.169	0.17	0.169	0.169	0.17	0.171	0.17	0.17	
ABS(A2)	0.309	0.307	0.307	0.303	0.306	0.304	0.309	0.307	0.305	0.267	0.841	
	0.308	0.308	0.309	0.307	0.304	0.308	0.308	0.307	0.308	0.278	0.845	
	0.306	0.309	0.31	0.309	0.308	0.31	0.301	0.303	0.301	0.277	0.847	
A2-A1	0.142	0.138	0.137	0.137	0.14	0.132	0.139	0.136	0.136	0.098	0.671	
	0.14	0.141	0.138	0.138	0.135	0.138	0.137	0.138	0.138	0.108	0.674	
	0.137	0.139	0.141	0.14	0.138	0.141	0.132	0.133	0.13	0.107	0.677	
Avg	0.139667	0.139333	0.138667	0.138333	0.137667	0.137	0.136	0.135667	0.134667	0.104333	0.674	
C (g/L)	0.020674	0.020092	0.019946	0.019946	0.020383	0.019218	0.020237	0.019801	0.019801	0.014268	0.097693	
	0.020383	0.020529	0.020092	0.020092	0.019655	0.020092	0.019946	0.020092	0.020092	0.015724	0.09813	
	0.019946	0.020237	0.020529	0.020383	0.020092	0.020529	0.019218	0.019364	0.018927	0.015578	0.098566	
Cavg	0.020334	0.020286	0.020189	0.02014	0.020043	0.019946	0.019801	0.019752	0.019607	0.010127	0.06542	
C (mg/L)	20.33447	20.28594	20.18888	20.14035	20.04328	19.94622	19.80063	19.7521	19.6065	10.12679	65.41972	
Original C(mg/L)	1020.768	1015.915	1006.208	1001.355	991.6491	981.9429	967.3837	962.5306	947.9713			55.29293

Experiment 3: 1000ppm solution concentration

Assay performed on March 24, 2007

	Initial	0min	10min	0min	10min	0min	10min	0min	10min	Blank	Control
ABS(A1)	0.185	0.182	0.18	0.179	0.181	0.185	0.181	0.182	0.184	0.183	0.202
	0.184	0.181	0.181	0.183	0.184	0.185	0.181	0.181	0.187	0.185	0.198
	0.187	0.181	0.181	0.181	0.183	0.184	0.183	0.184	0.186	0.184	0.198
ABS(A2)	0.342	0.339	0.337	0.335	0.331	0.334	0.322	0.328	0.329	0.216	0.827
	0.344	0.335	0.335	0.334	0.34	0.337	0.334	0.331	0.327	0.218	0.826
	0.341	0.337	0.333	0.333	0.332	0.333	0.334	0.327	0.334	0.215	0.827
A2-A1	0.157	0.157	0.157	0.156	0.15	0.149	0.141	0.146	0.145	0.033	0.625
	0.16	0.154	0.154	0.151	0.156	0.152	0.153	0.15	0.14	0.033	0.628
	0.154	0.156	0.152	0.152	0.149	0.149	0.151	0.143	0.148	0.031	0.629
Avg	0.157	0.155667	0.154333	0.153	0.151667	0.15	0.148333	0.146333	0.144333	0.032333	0.627333
C (g/L)	0.022858	0.022858	0.022858	0.022712	0.021839	0.021693	0.020529	0.021257	0.021111	0.004805	0.090996
	0.023295	0.022421	0.022421	0.021985	0.022712	0.02213	0.022276	0.021839	0.020383	0.004805	0.091432
	0.022421	0.022712	0.02213	0.02213	0.021693	0.021693	0.021985	0.02082	0.021548	0.004513	0.091578
Cavg	0.022858	0.022664	0.02247	0.022276	0.022082	0.021839	0.021596	0.021305	0.021014	0.003138	0.06089
C (mg/L)	22.85808	22.66395	22.46983	22.27571	22.08158	21.83893	21.59627	21.30509	21.0139	3.138335	60.89017
Original C(mg/L)	1971.974	1952.562	1933.15	1913.737	1894.325	1870.059	1845.794	1816.675	1787.557		

57.75183

Experiment 4: 2000ppm solution concentration

Assay performed on March 25, 2007

	Initial	0min	10min	0min	10min	0min	10min	0min	10min	Blank	Control
ABS(A1)	0.181	0.182	0.183	0.184	0.182	0.183	0.185	0.182	0.182	0.183	0.179
	0.18	0.183	0.182	0.184	0.184	0.183	0.185	0.181	0.182	0.184	0.181
	0.181	0.181	0.183	0.185	0.183	0.183	0.185	0.182	0.183	0.186	0.181
ABS(A2)	0.443	0.438	0.435	0.436	0.432	0.431	0.43	0.424	0.422	0.215	0.798
	0.441	0.437	0.437	0.437	0.432	0.432	0.429	0.426	0.42	0.214	0.797
	0.44	0.439	0.436	0.437	0.433	0.431	0.43	0.424	0.421	0.219	0.799
A2-A1	0.262	0.256	0.252	0.252	0.25	0.248	0.245	0.242	0.24	0.032	0.619
	0.261	0.254	0.255	0.253	0.248	0.249	0.244	0.245	0.238	0.03	0.616
	0.259	0.258	0.253	0.252	0.25	0.248	0.245	0.242	0.238	0.033	0.618
Avg	0.260667	0.256	0.253333	0.252333	0.249333	0.248333	0.244667	0.243	0.238667	0.031667	0.617667
C (g/L)	0.038145	0.037272	0.036689	0.036689	0.036398	0.036107	0.03567	0.035233	0.034942	0.004659	0.090122
	0.038	0.036981	0.037126	0.036835	0.036107	0.036253	0.035525	0.03567	0.034651	0.004368	0.089685
	0.037709	0.037563	0.036835	0.036689	0.036398	0.036107	0.03567	0.035233	0.034651	0.004805	0.089976
Cavg	0.037951	0.037272	0.036884	0.036738	0.036301	0.036156	0.035622	0.035379	0.034748	0.003074	0.059952
C (mg/L)	37.9512	37.27177	36.88352	36.73793	36.30115	36.15556	35.62172	35.37906	34.74816	3.073627	59.9519
Original C (mg/L)	3487.758	3419.814	3380.99	3366.43	3322.753	3308.193	3254.809	3230.544	3167.453		

56.87828