University of Southern Queensland
Faculty of Engineering and Surveying

Comparison of on-pond measurement and Back Calculation of Odour Emission Rates from Anaerobic Piggery Lagoons

A dissertation submitted by
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CERTIFICATION OF DISSERTATION

I certify that the ideas, experimental work, results, analyses and conclusions reported in this dissertation are entirely my own effort, except where otherwise acknowledge. I also certify that the work is original and has not been previously submitted for any other award, except where otherwise acknowledged.

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Signature of candidate Date

ENDORSEMENT

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Signature of supervisor Date

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Chapter 1 Introduction

1.1 Background

Odours are emitted from numerous sources and can form a natural part of the environment. The sources of odour range from natural to industrial sources and can be perceived by the community dependant upon a number of factors. These factors include frequency, intensity, duration, offensiveness and location (FIDOL). Or in other words how strong an odour is, at what level it becomes detectable, how long it can be smelt for, whether or not the odour is an acceptable or unacceptable smell as judged by the receptor (residents) and where the odour is smelt.

Intensive livestock operations cover a wide range of animal production enterprises, with all of these emitting odours. Essentially, intensive livestock in Queensland, and a certain extent Australia, refers to piggeries, feedlots and intensive dairy and poultry operations. Odour emissions from these operations can be a significant concern when the distance to nearby residents is small enough that odour from the operations is detected. The distance to receptors is a concern for intensive livestock operations as it may hamper their ability to develop new sites or expand existing sites.

The piggery industry in Australia relies upon anaerobic treatment to treat its liquid wastes. These earthen lagoons treat liquid wastes through degradation via biological activity (Barth 1985; Casey and McGahan 2000). As these lagoons emit up to 80% of the odour from a piggery (Smith *et al.,* 1999), it is imperative for the piggery industry that odour be better quantified.

Numerous methods have been adopted throughout the world for the measurement of odour including, trained field sniffers, electronic noses, olfactometry and electronic methods such as gas chromatography. Although these methods all have can be used, olfactometry is currently deemed to be the most appropriate method for

accurate and repeatable determination of odour. This is due to the standardisation of olfactometry through the Australian / New Zealand Standard for Dynamic Olfactometry and that olfactometry uses a standardised panel of “sniffers” which tend to give a repeatable indication of odour concentration. This is important as often, electronic measures cannot relate odour back to the human nose, which is the ultimate assessor of odour.

The way in which odour emission rates (OERs) from lagoons are determined is subject to debate. Currently the most commonly used methods are direct and indirect methods. Direct methods refer to placing enclosures on the ponds to measure the emissions whereas indirect methods refer to taking downwind samples on or near a pond and calculating an emission rate.

Worldwide the odour community is currently divided into two camps that disagree on how to directly measure odour, those who use the UNSW wind tunnel or similar (Jiang *et al.,* 1995; Byler *et al.,* 2004; Hudson and Casey 2002; Heber *et al.,* 2000; Schmidt and Bicudo 2002; Bliss *et al.,* 1995) or the USEPA flux chamber (Gholson *et al.,* 1989; Heber *et al.,* 2000; Feddes *et al.,* 2001; Witherspoon *et al.,* 2002; Schmidt and Bicudo 2002; Gholson *et al.,* 1991; Kienbusch 1986). The majority of peer reviewed literature shows that static chambers such as the USEPA flux chamber under predict emissions (Gao *et al.,* 1998b; Jiang and Kaye 1996) and based on this, the literature recommends wind tunnel type devices as the most appropriate method of determining emissions (Smith and Watts 1994a; Jiang and Kaye 1996; Gao *et al.,* 1998a). Based on these reviews it was decided to compare the indirect STINK model (Smith 1995) with the UNSW wind tunnel to assess the appropriateness of the methods for determining odour emission rates for area sources.

1.2 Objectives

The objective of this project was to assess the suitability of the STINK model and UNSW wind tunnel for determining odour emission rates from anaerobic piggery lagoons. In particular

- Determining if the model compared well with UNSW wind tunnel measurements from the same source;
- The overall efficacy of the model; and
- The relationship between source footprint and predicted odour emission rate.

Chapter 2 Literature review

2.1 Odour measurement

Olfactometry is the tool of choice for examining odour emissions and impacts. It makes use of the olfactory senses that are found in the human nose to determine odour concentrations using human panellists. Work undertaken, such as that of Gralapp *et al.* (2001) shows that the human nose still outperforms current instrumental methods. This is because the available methods (eg. gas chromatography, electronic noses) are unable to match the interaction between numerous odorous compounds from a source and relate these to a person's sense of smell.

2.1.1 Olfactometry

Olfactometers operate by passing a diluted sample of odour to a trained panel of people (sniffers or panellists) who then rate the odour based on whether they can detect the odour (certain), think they can detect the odour (inkling) or are simply guessing. Forced choice refers to the panellists having to provide a response even if they cannot detect odour. The concentration presented to the panellists is increased by doubling the concentration until each panellist can detect the odour with certainty.

There are a number of olfactometry standards however, in general, they vary with respect to panel selection and the process used to calculate results. In the past a number of olfactometry standards have been used in Australia and overseas (Watts 2000), including :

- Queensland Department of Environment Method 6;
- Victorian EPA Method;
- Dutch Draft Standard NVN2820;
- Australian Standard (AS4323.3); and

- CEN Standard (CEN-TC264).

Watts (2000) found that the major difference between European and American standards for olfactometry were;

- Volumetric flow rate;
- Face velocity;
- Panel selection;
- Instrument calibration (American standards do not require); and
- Guessing versus certain and correct criteria.

Until recently, the most commonly used standard for olfactometry in Australia was the draft Dutch standard (NVN). With the publication of Australian and New Zealand standard (AS4323.3) in (2001) the NVN standard is redundant.

The CEN and the Australian standard are essentially the same. The major difference between them is the number of times a series of dilutions (rounds) going from low to high concentration are presented to the panellists. The Australian standard states that, "*A preliminary round of a measurement shall be made and the data systematically discarded (i.e. it is always excluded)*" (Standards Australia 2001). Whereas the CEN standard states that, "*A preliminary round of a measurement may be made and the data systematically discarded*" (CEN 1999).

Simply put, under AS4323.3, three sample rounds are presented to the panel with the first being discarded. Whereas under the CEN standard, odour determination can be made using first two rounds.

Jiang (2002) reported confusion when attempting to follow the CEN and the Australian Standards. He found that olfactometry concentrations could be subject to the operator's personal view. It should be noted that Jiang followed the CEN standard and as such did not comply with the Australian Standard as defined in his project report. Jiang (2002) preferred to present the odour to his panellists at a

preselected concentration, if the odour was detected by the panellist, he then would dilute the odour until it could not be detected, this being the start of his first round.

Experience with olfactometry has shown that, often the first round has a different geometric mean when compared to the following rounds (i.e. rounds 2 and 3). The author has a number of theories as to why this occurs. The most plausible reason is that after the butanol screening at the start of the session the panellists are still of the belief that are looking for n-butanol. The panellist does not detect the new odour until they can respond with certainty and correctly. Once the new odour has been detected, the panellist can more easily determine a difference in the following rounds, perhaps due to some kind of “memory training”. After they have smelt the new sample, they can then repeatedly detect it for the rest of the presentations. A number of commercial laboratories give the panellist a smell of the concentrated odour prior to analysis to help them recognise odours that are different to n-butanol.

The Australian Standard for olfactometry has specific details covering accuracy, instability and repeatability. Accuracy and instability are calculated by using carbon monoxide (CO) as a tracer gas. The accuracy clause refers to closeness of agreement between test result and the accepted reference value with the instability criterion relating to the change of a characteristic over a stated period.

Carbon monoxide is used to confirm the dilutions within the olfactometer. For each dilution step a known concentration of CO is passed through the olfactometer at a set dilution. Generally, an amount of time is required for the odorous (or CO) air to mix with odour free air and to travel to the panellists this is defined as a settling time. After the settling time has passed, ten concentration readings are taken five seconds apart. Once this is complete, the olfactometer is flushed with clean air and the process is repeated four more times.

From this data, the instrumental accuracy (as a function of the standard deviation of instability) and the accuracy (closeness to the theoretical value) are calculated. The

instability (I_d) should be less than 5% and the accuracy (A_d) should be less than or equal to 0.2 (Standards Australia 2001).

The Australian olfactometry standard states that, for an olfactometer complying with the standard, two tests performed on the same testing material in one laboratory under repeatability conditions will not be larger than a factor of 3 in 95% of cases. Sneath and Clarkson (2000) and the United Kingdom Environment Agency (2002) showed that for a sample at a given concentration, the variability attributable to olfactometry decreases as the number of samples analysed increases. A graphical representation of the decrease in variability associated with a hypothetical sample of 100 OU is shown in Figure 1. This data assumes that the olfactometer only just complies with the requirements of the standard (AS4323.3).

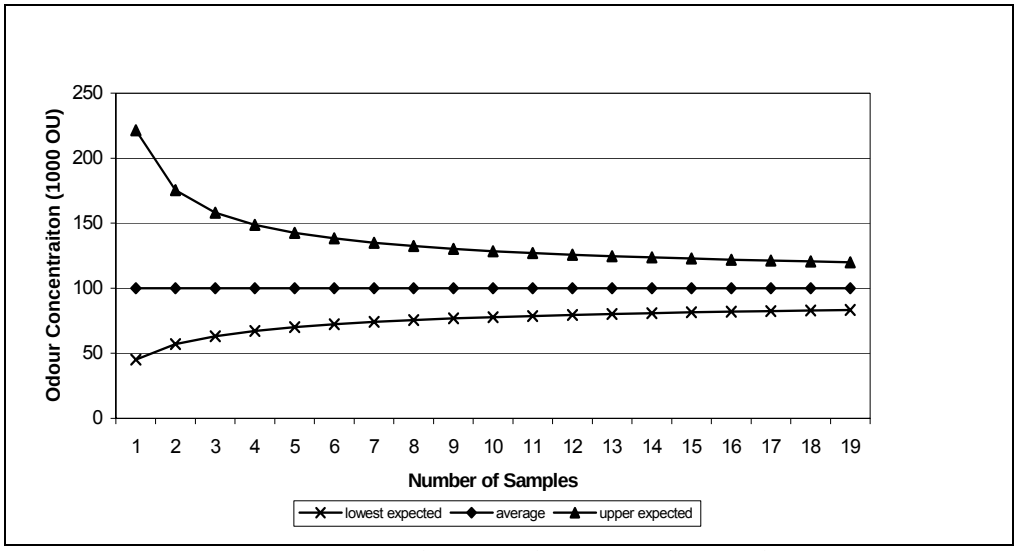

Figure 1: Upper and lower bounds for a 100 OU sample for an olfactometer meeting the minimum requirements of AS4323.3

2.1.2 Olfactometry units

There has been conjecture over the appropriate units to use when reporting odour concentrations. The Australian Standard for olfactometry (Standards Australia 2001) uses the odour unit (OU) as its unit, whereas the CEN standard (CEN 1999) upon which the Australian Standard is based uses odour units per cubic metre (OU/m^3). By

definition, one odour unit is the amount of (a mixture of) odorants present in one cubic metre of odorous gas (under standard conditions) at the panel threshold. In practical terms, the number of odour units that a sample is, refers to the average of the diltions at which a group of panellists are able to determine with certainty that the odour is present when presented to the panellists in the olfactometer.

When undertaking emission calculations, the units play an important role for resultant emission rate. The use of OU/m^3 is thought to be relating back to classical units such as an amount of compound per m^3 of air (eg. $\mu g/m^3$, mg/m^3). There is not really a certain amount of odour in a cubic metre as it is a dilution ratio rather than a true concentration. The use of OU/m^3 when placed into the continuity equation results in an emission rate in $OUm^{-2}s^{-1}$ whereas the use of OU results in OU.m/s. Both of which are correct depending upon which standard you reference.

A number of researchers have reported odour emission units in $OUm^{-2}s^{-1}$(Guo *et al.,* 2001; Heber *et al.,* 2000; McGahan *et al.,* 2001; Sarkar and Hobbs 2003; Witherspoon *et al.,* 2002) and others have reported odour emission units in $OUms^{-1}$ (Hudson and Casey 2002; Smith and Watts 1994b). An alternative to these emissions which has been published by Wood *et al.* (2001) and Gulovsen *et al.* (1992) is $OU.m^3m^{-2}s^{-1}$. These units seem to be a combination of the two units previously stated.

For the purposes of this project, the use of OU/m^3 for concentrations and $OUm^{-2}s^{-1}$ has been adopted for emissions. These however can be interchanged with the other units listed above.

2.1.3 Olfactometry analysis

Olfactometry should be undertaken to meet the requirements of the Australian / New Zealand Standard for Dynamic Olfactometry (AS4323.3) (Standards Australia 2001) or alternatively prEN 13725 (CEN 1999, ASSTM 1991).

2.1.3.1 Panellist screening

Olfactometers use specially selected panellists with each panellist being screened with the reference gas, n-butanol, prior to them undertaking odour analysis. The screening is used to select panellists that can detect n-butanol repeatedly at a concentration between 20 and 80 ppb. Only a panellist with a long-term (10 sample) n-butanol history within this range can be used.

2.1.3.2 Sample analysis

Odorous air is diluted and presented to the panellists in one of three available ports with the other two ports emitting odour free air. The panellists are then asked to determine if they can detect a difference between the three ports over a period of 15 seconds after which the odorous air stops. The panellists then must respond whether they are certain, uncertain or guessing from which port the odour was emitted.

This process is repeated by doubling the strength of the previous presentation until all panellists had responded with certainty and correctly for two consecutive presentations. The panellists' individual threshold estimate (Z_{ITE}) are then determined by calculating the geometric mean of the dilution at which the panellists did not respond with certain and correct and the first of the two dilutions where the panellists responded with certain and correct. This dilution series is defined as a round. Three rounds should be undertaken for each sample to meet the standard.

At the end of the three rounds, the results from the first round are discarded (Standards Australia 2001). The individual threshold estimates from rounds two and three are then geometrically averaged ($\overline{Z}_{ITE}$). The ratio between $\overline{Z}_{ITE}$ and Z_{ITE} is defined as ΔZ. The calculation of ΔZ is as follows in Equation 1 and Equation 2.

If $Z_{ITE} \geq \overline{Z}_{ITE}$ then $\Delta Z = \frac{Z_{ITE}}{\overline{Z}_{ITE}}$ Equation 1

If $Z_{ITE} \leq \overline{Z}_{ITE}$ then $\Delta Z = \frac{\overline{Z}_{ITE}}{Z_{ITE}}$ Equation 2

If ΔZ is greater than ± 5 then all ITEs of the panel member with the largest ΔZ are excluded from the data set. The screening procedure is then repeated, after re-calculation of $\overline{Z}_{ITE}$ for that measurement. If panel member(s) again do not comply, the panel member with the largest ΔZ is omitted. This is repeated until all panel members in the dataset comply. A minimum of four panellists must remain after the above screening procedure for the analysis to comply with the standard. The last value of $\overline{Z}_{ITE}$ is then defined as the odour concentration and expressed as odour units (OU or OU/m^3).

2.1.4 Assessment of olfactometer variation

The variation in olfactometry results can be assessed by calculating the 95% confidence interval on the number of samples with the worst case repeatability standard deviation (s_r) of 0.1726 as detailed in the standard. For an olfactometer exceeding the minimum requirements of the standard, the confidence interval would be less than one for an olfactometer meeting the minimum requirements. For a given group of samples the average concentration is determined. This value is then placed into Equation 3 which provides an upper and lower confidence interval. This process has been successfully used by Sneath and Clarkson (2000) and Gostelow *et al.* (1993).

$$\bar{y}_w - t \times \frac{s_r}{\sqrt{n}} \leq m \leq \bar{y}_w + t \times \frac{s_r}{\sqrt{n}}$$ Equation 3

Where:

- $\bar{y}_w$ is the average of the odour concentration measurements;
- *m* is the expected odour concentration;

- *t* is the Student's t-factor for n samples (for n=∞, t=2 (Standards Australia 2001);
- *n* is the number of samples;
- $\sqrt{n}$ is the square root of the number of samples; and
- s_r is the repeatability standard deviation.

2.1.5 Sample bag material

A number of materials including Teflon, Tedlar and PET (Melinex and Nalophan) are listed in the Standard as odour free and suitable for use in odour sampling. The variety of suitable materials has seen a number of laboratories adopt different bag materials. For example, the UNSW odour laboratory uses Tedlar; The Odour Unit in Sydney uses Nalophan, The Department of Primary Industries and Fisheries in Queensland use Melinex and Ton van Harreveld (Odournet - Europe) use Nalophan.

2.1.6 Odour decay

When undertaking odour sampling, it is important to assess the suitability of the sample bag materials used for odour sampling, as odour samples stored in bags may decrease or increase with time. These changes may result in spurious emission rates being developed.

A recent study by van Harreveld (2003) showed that odour from a tobacco processing factory decayed after 12 hours of storage in a Nalophan sample bag. Based on this work he concluded that Nalophan was the most suitable for sampling environmental odours. In contrast to this was the work of Pollock and Friebel (2002) which concluded that Tedlar was more suitable for sampling than Nalophan for poultry odour.

The finding of van Harreveld was supported by a recent paper which showed Tedlar had a detectable odour concentration related to its manufacturing and required pre-treatment of the bag prior to odour sampling (Parker *et al.,* 2003). Koziel *et al.* (2004),

has supported the use of Melinex/Nalophan as they concluded that Melinex had better gas storage and recoveries than other bag materials such as Tedlar and Teflon.

Australian pork project 1628, *The effect of loading rate and spatial variability on pond odour emission rates* (See Hudson *et al.* (2004)), examined blank odour samples and sample decay for piggery odour samples. The project found that when odour free air was stored in a Melinex sample bag the measured odour concentrations were similar to the theoretical lower detection limit (LDL = 6 OU) of the olfactometer.

In the same project, an odour sample was analysed over a 24-hour period. The factor of 3 (worst case value for an olfactometer just complying with AS4323.3) for six samples was used in a calculation to determine the repeatability. This was undertaken in a similar fashion to that described by Sneath and Clarkson (2000). The results from this assessment are shown in Table 1 and Figure 2.

Table 1: Results of 95% confidence interval for odour samples

Range	Concentration (OU)
Calculated lower value	62
Average odour concentration	86
Calculated upper value	119

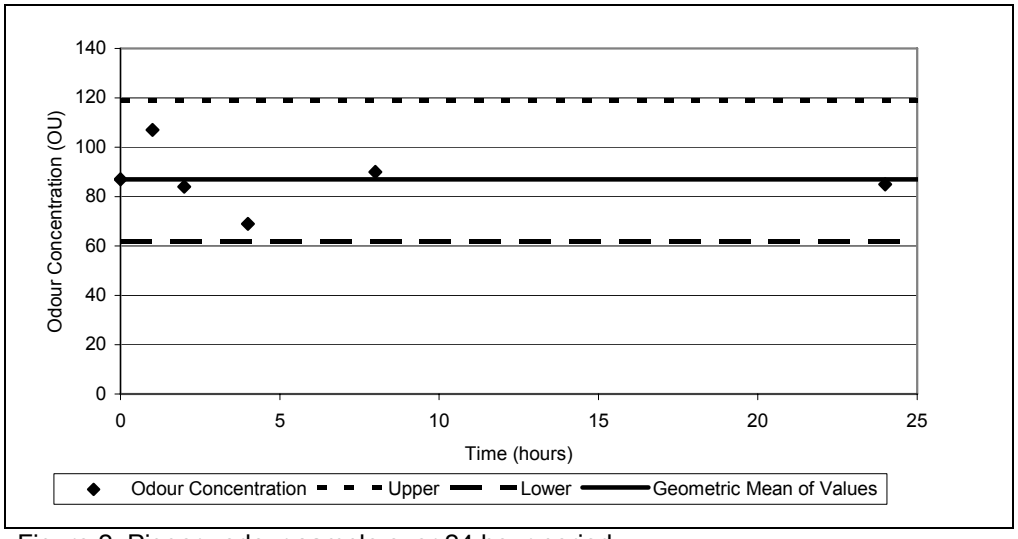

Figure 2: Piggery odour sample over 24 hour period

Figure 2 shows:

- The average odour concentration over the 24 hours was 86 OU; and
- All analyses undertaken over the 24 hour period provided odour concentrations that were within the range that could be associated with olfactometry.

Therefore it can be concluded that the piggery odour did not vary significantly over the 24 hour storage period when stored in Melinex bags. The interpretation of these results were independently confirmed by van Harreveld (2003), who recommended that samples could be stored for periods up to a maximum of 24 hours but ideally should be analysed with 12 hours when using Nalophan bags (Melinex).

2.2 Review of area source emission measurement

2.2.1 Introduction

Area sources, including piggery ponds, are the most difficult source from which to estimate odour emission rates. This is due to the fact that there is no way of directly measuring or sampling the emission. Therefore, indirect methods must be used where the emissions are sampled after they have mixed with the air stream (Watts

2000). Currently a number of methods are available to determine emissions from area sources; however, it is important to note that in these techniques there are a number of inherent uncertainties.

Watts (2000) broke the techniques into two areas, physical and downwind. Physical methods involve placing open bottomed enclosures over the emitting surface and determining the concentration of odour in the air exiting the enclosure. The second technique involves back calculating emissions from downwind concentrations (Sommer *et al.,* 2004; Sarkar and Hobbs 2003; Smith 1995; Smith and Kelly 1996; Wilson *et al.,* 1983; Turnbull and Harrison 2000; Zahn *et al.,* 2002; Rege and Tock 1996). Physical methods include wind tunnels / hoods (Heber *et al.,* 2000; Jiang *et al.,* 1995; Ryden and Lockyer 1985) and flux chambers (Gholson *et al.,* 1989; Feddes *et al.,* 2001; Martins 2000b; Wang *et al.,* 1997; Martins 2000a). Jiang and Kaye (1996) identified the wind tunnel method as a dynamic method where the supply air has a much higher flow rate than the other dynamic methods such as the USEPA flux chamber, which has a low carrier gas flow rate.

Of the methods currently available, the most commonly used methods in Australia are the UNSW wind tunnel and USEPA flux chamber. The flux chamber is typically used for the measurement of volatile organic compounds (VOCs) from soils (Martins 2000b; Stavropoulos *et al.,* 2002) whereas wind tunnels are predominantly used for sampling odour emissions (for more information see Heber *et al.* (2000), Jiang (2002), Jiang *et al.* (1995), Schmidt *et al.* (1999) and Smith and Watts (1994b)).

Downwind methods are used to calculate emission rates from emitting surfaces by combining a concentration measured downwind of the source with local meteorological conditions at the time of sampling. While a number of methods have been used, each method is based upon the premise that a downwind concentration can be related to an upwind source via a mathematical formula (Turnbull and Harrison 2000; Edgar *et al.,* 2002; Koppolu *et al.,* 2002; Sarkar and Hobbs 2003;

Smith 1993; Smith and Kelly 1996). Table 2 shows the review of wind tunnel and back-calculation methods undertaken by Smith and Kelly (1995).

Table 2: Difference between wind tunnel and back calculation methods

Wind Tunnel	Back-calculation
• sample emission rate from a small area of source • equipment intensive – requires wind tunnel, fan and filter, sampling equipment and anemometers • emissions are mixed with ambient air in a bounded air flow • representative sample of air/odour mixture taken at downwind end of tunnel (complete mixing is assumed) • odour concentration time invariant therefore sample time or rate unimportant • wind speed over the sample area is controllable • vertical wind speed profile over the sample area is a function of the tunnel geometry and the aerodynamic roughness of the surface • emission rate calculated from measured concentration by a simple continuity equation • gives the emission rate corresponding to the average or bulk wind speed in the tunnel (wind speed or emission rate can be adjusted to the ambient value)	• samples 'average' emission rate from a relatively large area of source • less equipment intensive – requires only sampling equipment and anemometers • emissions are mixed with ambient air in an essentially unbounded air flow • sample of air/odour mixture taken at a known height at any selected receptor location downwind of source • odour concentration time variant due to turbulence therefore sampling time must correspond to the averaging time used for the wind speed measurements • wind speed over source is not controllable • vertical wind speed profile over source is a function of atmospheric stability and the aerodynamic roughness of the surface • emission rate calculated from measured concentration by a simple continuity equation disguised as a dispersion model gives the actual emission rate for the prevailing ambient conditions

2.2.2 Emission theory

The classic theory for describing volatile emissions from a liquid surface is the two film theory (Gholson *et al.,* 1991). For volatile compounds where Henry's law constant is greater than $1x10^{-3}$ atm m^3 mol^{-1}, the liquid film resistance is the controlling factor for emission (Gholson *et al.,* 1991). Liquid turbulence and diffusion have been shown to be the environmental factors controlling the liquid mass transfer coefficient (Gholson *et al.,* 1991). These factors are controlled by surface wind velocity and surface temperature (Gholson *et al.,* 1991; Martins 2000b).

Generally as turbulence across a surface increases, compounds stripped to the atmosphere increase (Sattler and McDonald 2002). Unfortunately, with the numerous compounds contained in odours, it becomes difficult to assess the boundary layer conditions (Martins 2000b).

2.2.3 Emission rate calculation

The emission of a compound or odour from a wind tunnel or flux hood can be described by a continuity equation (Smith and Watts 1994a). Odour emission rate (E) is a function of the concentration exiting the unit (C), the flow rate inside the unit (Q) and the surface area covered by the unit (A_S). The continuity equation is shown in Equation 4.

$$E = \frac{C \times Q}{A_S} \qquad \text{Equation 4}$$

For wind tunnels the formula varies slightly as it takes into account the internal face of the tunnel and the footprint (area enclosed) of the tunnel. This is shown in Equation 5 (Smith and Watts 1994a).

$$E = C \times V_t \left(\frac{A_t}{A_S} \right) \qquad \text{Equation 5}$$

Where:

- V_t is the bulk wind speed of the tunnel (m/s);

- A_t is the cross sectional area of the tunnel (m^2); and
- A_S is the surface area covered by the tunnel (m^2);

2.2.4 Isolation (flux) chamber

An isolation chamber is an enclosure of known volume used for direct measurement of emissions from solid and liquid surfaces. The chamber has an open end that is placed on a surface to quantify emissions (Martins 2000b). Typically, isolation chambers have a much lower airflow rate than wind tunnels. Kienbusch (1986), Feddes *et al.* (2001), Gulovsen *et al.* (1992), Gholson *et al.* (1989; 1991), Torre (1991) and Martins (2000b; 2000a) have detailed the use of various styles of flux chamber for the measurement of emissions from solid and liquid sources. A schematic of the USEPA flux chamber as described by Kienbusch (1986) is shown in Figure 3.

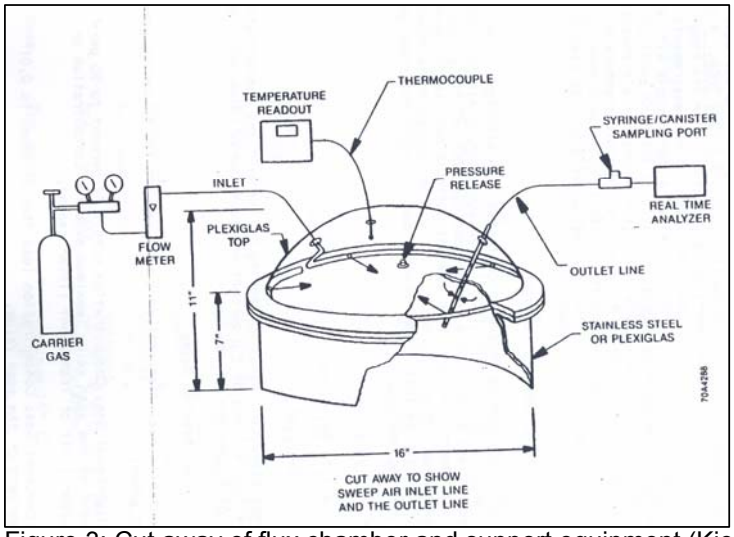

Figure 3: Cut away of flux chamber and support equipment (Kienbusch 1986)

Kienbusch (1986) identified the flux chamber as a promising technique for measuring gas emission rates from land surfaces and had a number of benefits including:

- a low detection limit;
- easily obtained accuracy and precision data;
- simple and economical equipment requirements;

- minimal time requirements; and
- rapid and simple data reduction and the applicability to a wide variety of surfaces.

The chamber operates by passing sweep air (instrument air or nitrogen) through the device at a rate of 5 litres per minute (Kienbusch 1986). It has been shown that flux chambers derive lower emission rates than wind tunnels or back calculation methods. Gao and Yates (1998b) examined two closed chambers to estimate the emission rates of a soil fumigant. They found that the two chambers (similar to the USEPA flux chambers) underestimated emissions because of the low flush rates. The design of the chambers meant that the concentration of material in the chamber volume increased with time, which in turn depressed the concentration gradient across the soil-air interface, this in turn decreased emissions. In their study, Jiang and Kaye (1996) examined a UNSW wind tunnel and a USEPA flux chamber and made similar conclusions to Gao and Yates, that flux chambers underestimate emissions and that the flux chambers exhibit gas phase controlled volatilisation.

Other more recent research by Sarkar and Hobbs (2003) and Sommer *et al.* (2004) have compared back calculation methods with wind tunnels and flux chambers respectively. Whilst Sarkar and Hobbs (2003) found a good relationship between a wind tunnel and their wind tunnel technique, Sommer *et al.* (2004) found that their large flux chamber predicted emission rates that were 12-22% of that measured with the back calculation techniques. Gao and Yates (1998b) and Jiang and Kaye (1996) both concluded that emissions could be suppressed by covering an emitting surface and limiting airflow over the surface.

2.2.5 Wind tunnels

2.2.5.1 Background

A wind tunnel is a portable enclosure that is placed on a surface that can be either liquid or solid. Air (carbon filtered or ambient) is blown or sucked (dependant upon

the design) through the tunnel to simulate the transport of odorous compounds by the local meteorological conditions. In their review of sampling techniques, Smith and Watts (1994a) concluded that wind tunnels were the best method for obtaining emission rates from area sources. Recent work in the United States (eg. Heber *et al.* (2000), Byler *et al.* (2004) and Schmidt and Bicudo (2002)) has favoured the use of wind tunnels for estimating odour emissions from area sources, thus, indicating acceptance in Australia and the United States of wind tunnels for determining area source emission rates.

Wind tunnels have been used extensively to measure emission rates from area sources. Recent studies also include those of Smith and Watts (1994b), Hudson and Casey (2002), Gwynne *et al.* (2002) and Ryden and Lockyer (1985). It should be noted that the types of tunnels studied by Smith and Watts (1994b) are no longer in use in Australia since the development of the more modern UNSW wind tunnel.

Wind tunnels are generally of similar design but have different operating conditions. The differences can include whether the wind is blown or drawn through the chamber and the use of carbon filtered air or ambient air. Smith and Watts (1994b) found that normalising data obtained from wind tunnels of two different sizes gave similar odour emission rates. The method of normalising used was a function of the different velocity profiles in the two tunnels. These velocity profiles depended heavily on the aerodynamic roughness of the feedlot surface (Smith and Watts 1994b).

Emission rates derived from wind tunnels are standardised using Equation 6 (Smith and Watts 1994b). This equation converts the wind tunnel emission rate at the sampling wind speed to an emission rate at 1 m/s.

$$\frac{E}{E_1} \approx V_t^{0.65} \qquad \text{Equation 6}$$

Where:

- E_1 and E are the standardised and calculated emission rates respectively; and
- V_t is the average wind speed in the wind tunnel.

Pollock (1997) detailed changing the factor of 0.65 which was derived from studies on feedlot pads (Equation 6) to 0.5 for use on pond surfaces based on the work of Bliss *et al.* (1995). Recent research by Schmidt *et al.* (1999) found that the exponent for odour on a lagoon surface was 0.89 (R^2 = 0.56) for odour and 0.90 (R^2 = 0.87) for hydrogen sulphide. Schmidt *et al.* (1999) hypothesised that the difference between their work and that of Smith and Watts (1994b) could be a result of one or more factors that included the type of odour source, surface roughness, wind tunnel design and the range of wind speeds tested.

2.2.5.2 University of New South Wales wind tunnel

The design of tunnel that is commonly used in Australia is the University of New South Wales (UNSW) wind tunnel (Figure 4) which is based on the Lindvall hood design. The development of the tunnel was described by Jiang *et al.* (1995). The UNSW wind tunnel is normally operated at an internal velocity of 0.3 m/s which was found to be the optimal velocity for use (Bliss *et al.,* 1995). However the operating range of the wind tunnel has been reported to be between 0.3 and 0.6 m/s (Bliss *et al.,* 1995).

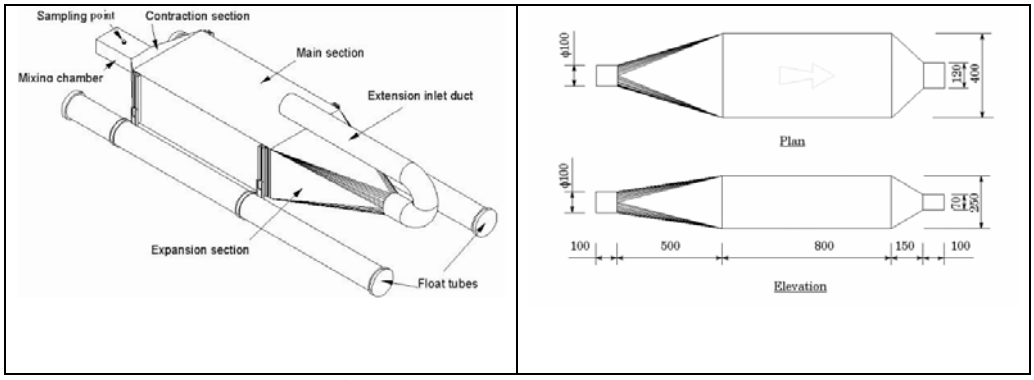

Figure 4: University of New South Wales wind tunnel (http://www.odour.unsw.edu.au)

Prior to design of the UNSW wind tunnel, the design of the Lindvall hood was evaluated to give a starting point for designing a new tunnel. This evaluation showed that flow straightening would be required to improve aerodynamic performance.

Jiang *et al.* (1995), described the design process of the UNSW wind tunnel. Their aim was to design a system, which had better aerodynamic performance than the Lindvall hood, and to find the relationship between air velocity and odour emission rate. The design of various wind tunnels was reviewed by Watts (2000) and further details of the Lindvall hood can be found in his review.

After initial testing of the new design, the flexible inlet duct was found to cause an uneven velocity distribution within the tunnel. To improve this uneven distribution within the tunnel, a fixed extension duct was designed to enable a consistent velocity profile when air entered the expansion section of the tunnel (Jiang *et al.,* 1995). In addition to the inlet duct three vertical flat planes were placed with equidistant spacing between the walls to flatten the horizontal velocity distributions and to control the jet effect[1] (Jiang *et al.,* 1995). These changes are shown in Figure 5.

[1] Air was found to go straight through the hood and create a rotation inside the hood and a dead zone near the corners of the hood.

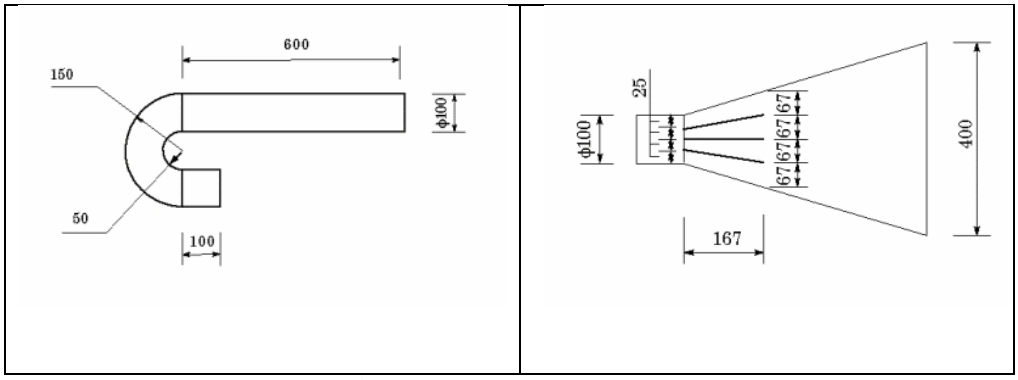

Figure 5: Extension duct and flat vanes (http://www.odour.unsw.edu.au)

After this work had been completed it was noted by the developer that even with the extension inlet duct there was still a significant amount of rotation of the flow within the tunnel. To negate the rotation effect a 10 mm thick perforated Teflon baffle was installed between the inlet duct and the main section. The Teflon baffle is shown in Figure 6.

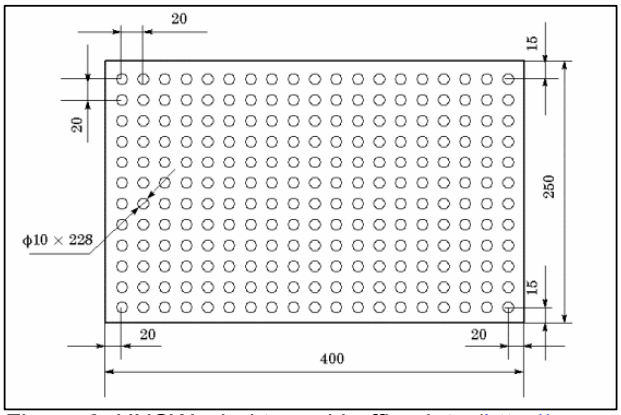

Figure 6: UNSW wind tunnel baffle plate (http://www.odour.unsw.edu.au)

Recent work by Baldo (2000) on the UNSW wind tunnel found that whilst the developers of the tunnel had attempted to remove the “jet effect” they have only managed to reduce it.

2.2.5.3 Recent developments with the UNSW wind tunnel design

Loubet *et al.* (1999a; 1999b) reviewed conventional wind tunnels and found that volatilisation fluxes could be biased because of the methods used to sample the air velocity and concentration in the duct. They found recovery rates varied between

70% and 87% depending on the flow rate inside of the tunnel (Loubet *et al.,* 1999a). This work identified two main causes of error associated with wind tunnel use:

- non-uniformity of the concentration profile in the measurement section leading to a underestimation of the flux of 11%; and
- non-uniformity of the wind speed profile leading to an overestimation of the flux of about 3%.

Subsequently the UNSW tunnel was modified based upon the studies undertaken by Loubet *et al.* (1999a; 1999b). The modification as published by Wang *et al.* (2001) is shown in Figure 7. Samples are drawn from the centre of the cross that is located in the exit of the extension (bottom of Figure 7).

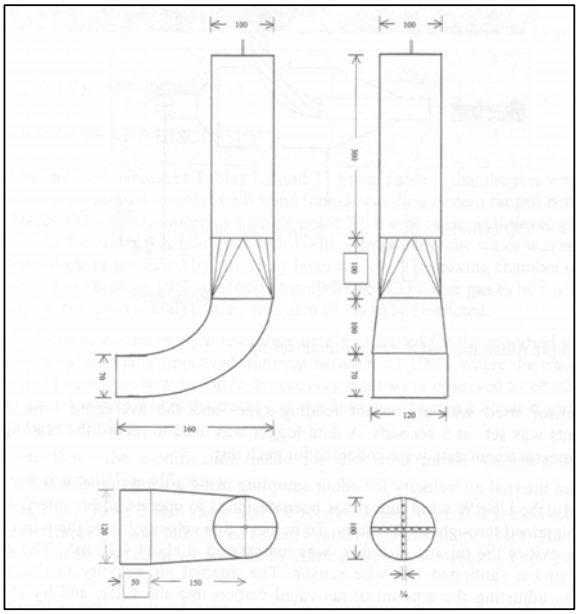

Figure 7: Modification to improve sampling efficiency

2.3 Dispersion and back calculation of odours

2.3.1 Background

Dispersion models are used worldwide for numerous purposes, primarily regulatory assessments. For regulatory assessments, emission rates for emitting sources are used in the models to predict concentrations in ambient air away from the source.

Of the methods available to determine odour emission rates from area sources, back calculation is probably the most cost effective. Subsequently the back calculation of emissions from area sources has been widely reported as an alternative to direct measurement techniques.

The advantage offered by back calculation is that it can provide a spatially averaged emission rate (Harris *et al.,* 1996; Smith and Kelly 1996) rather than a single point sample of a source based upon the limited footprint of an enclosure technique. A number of assumptions must be made when using Gaussian (or to a certain extent other) models to model odour emissions. The assumptions detailed by Harris *et al.* (1996) include:

- cross-sections through the plumes have a Gaussian or normal distribution over the time period;
- the pollutant being dispersed obeys the law of conservation of mass;
- the ground is a barrier to vertical mixing and is represented as a flat surface reflecting the plume back into turbulent air flow;
- odour is neutrally buoyant – This assumption implies that the temperature at which the odour is released equals the ambient air temperature;
- the odour does not degrade after release (conservation of mass);
- the emission height at the source is equal to the roughness length of the surface;
- odour source emissions do not vary with time; and
- odour source emissions do not vary spatially;

Dispersion is the overall description that can be used to describe odour moving away from a source and refers to the spreading or diffusion of the odour. It should be noted that diffusion would be more of a factor during calm conditions whereas dispersion would be dominant during turbulent conditions. When examining the dispersion of

odour from a source a number of meteorological and local factors affect the dispersion.

2.3.2 Atmospheric stability

Atmospheric stability for dispersion modelling is generally defined through Pasquill stability classes (Beychok 1994; PAE 2003a). The stability classes are used to represent the inclination of the atmosphere to resist or enhance upward movement of an odour. Pasquill classes range from A-F, with A being very unstable and F being very stable. A stability class of C is considered neutral. Normally E and F are only seen at night.

A stable atmosphere can be defined as one in which a small parcel of air given an upward displacement would tend to return to its original position. Neutral stratification refers to a case where there is no tendency for the pollutant to be displaced either up or down from its original position and an unstable atmosphere exists when a small parcel of air continues to rise after being given as small displacement upwards (Harris *et al.,* 1996). A description of the stability classes as discussed by Harris *et al.* (1996) is shown in Table 3.

Table 3: Description of stability classes

Pasquill Class	Description
A	Extremely unstable
B	Moderately unstable
C	Slightly unstable
D	Neutral
E	Slightly stable
F	Moderately stable

2.3.3 Wind speed and surface roughness length

Wind speed is an important factor when modelling as it has two effects. Firstly it is a direct input into modelling and influences the distance travelled by the plume.

Secondly, it is an input for calculating stability classes that are also a model input. Wind speed does not remain constant as height changes. The effect of altitude on velocity involves two factors (Beychok 1994):

- Degree of turbulent mixing as characterised by stability class; and
- The surface roughness, which induces friction.

As a rule of thumb, the more even the terrain (lower roughness) the steeper the profile (Harris *et al.,* 1996). The wind speed at a particular height can be calculated from wind speed measured on-site using the power law, which assumes that the wind speed increases logarithmically with height (Beychok 1994; USEPA 2000a; VicEPA 2000). The power law is shown as Equation 7. An example of the power law for 2 m/s at a sampling height of 2 metres is shown in Figure 8.

$$U_z = U_{ref}\left(\frac{Z}{Z_{ref}}\right)^a \qquad \text{Equation 7}$$

Where:

- U_z (m/s) is the wind speed at a height of Z (m) above the ground;
- U_{ref} (m/s) is the wind speed at anemometer height Z_{ref} (m); and
- *a*, the wind profile exponent, is a function of the stability class, surface roughness and height and is detailed in the literature (see USEPA (2000a)).

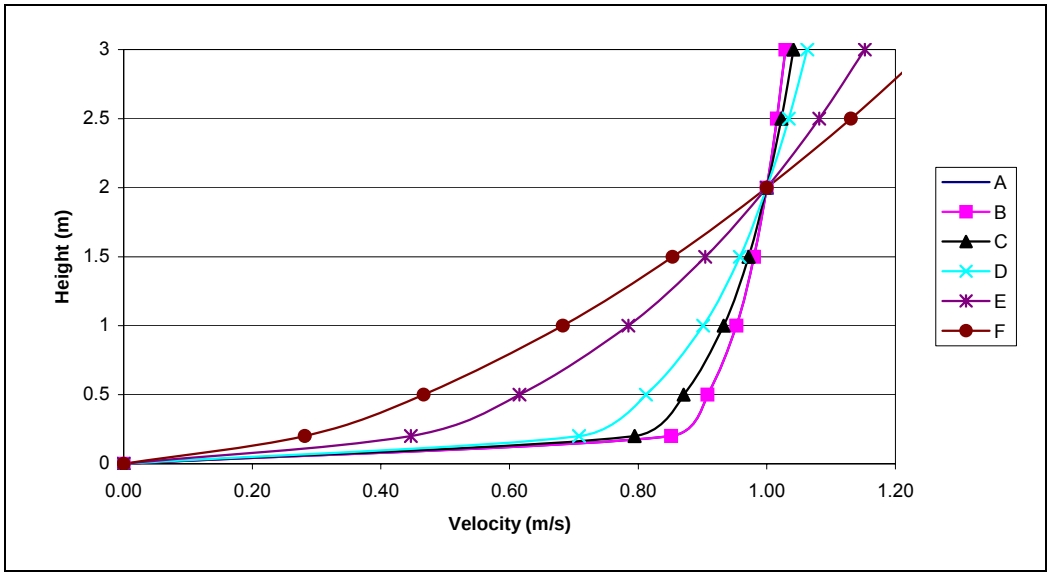

Figure 8: Example of power law for 2 m/s at 2 metres

2.3.4 Turbulence

The primary factor determining the dispersion of gases is atmospheric turbulence (Beychok 1994; Harris *et al.,* 1996). Harris *et al.* (1996) broke down turbulence into two categories 1)friction (mechanical turbulence) and 2) convection (thermal turbulence).

Mechanical turbulence is a result of a parcel of air passing over a surface near the ground. It will be affected by features such as vegetation and structures. Increasing horizontal wind speed near the ground surface cannot overcome the surface frictional effects and a result of this, eddies form and drift upwards, increasing the depth of turbulent flow (Harris *et al.,* 1996). Thermal turbulence is a result of differences in the surface and air temperatures creating convective currents (Harris *et al.,* 1996). High turbulence generally equates to unstable conditions and low turbulence equates to stable conditions.

2.3.5 Source footprint

Smith (1995) speculated, that each emitting point on a source would contribute equally to the resultant downwind concentration, provided the sample point was far

enough away from the source. The footprint refers to the areal source defined in the STINK model. Beychok (1994) showed that the horizontal dispersion, under different stability conditions, was minimal between 10 and 100 metres from a source. Thus it would be expected that source footprint would be an issue for samples collected within 100 metres of a source. Unfortunately, no published data is available on this subject for odour. A visual representation of the calculated horizontal dispersion under various atmospheric stability conditions is shown in Figure 9.

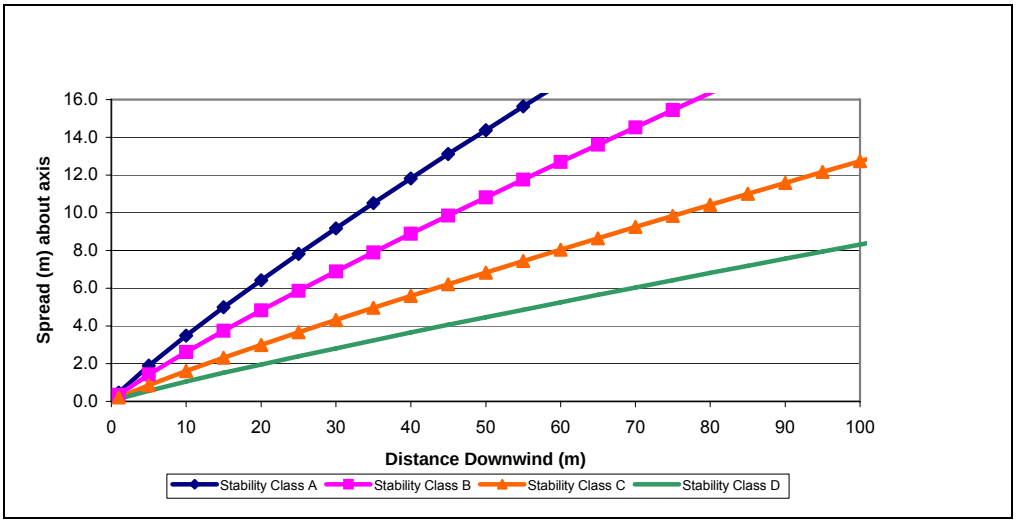

Figure 9: Horizontal spread according to distance from source

2.4 Review of back calculation methods

Koppolu *et al.* (2002) identified a number of factors, which make the assessment of odour impacts challenging. The most important of these, was that odour perception by the human nose cannot be assessed by current instrumental techniques. Assumptions such as homogenous emissions from sources have also been found to be important (Sarkar and Hobbs 2003). Factors such a these, should be taken into account when utilising dispersion modelling for impact or emission rate predictions.

Smith and Watts (1994a) reviewed methods for the indirect estimation of emission rates. These methods included the mass balance method, the energy balance method, the theoretical profile shape (TPS) method and the Gaussian plume. In

addition to these, a number of other models have recently been developed, including the Eulerian model of Sarkar and Hobbs (2003) and backward Lagrangian stochastic (bLS) models of Sommer *et al.* (2004) and Flesch *et al.* (2005).

2.4.1 Mass balance method

The mass balance method involves the simultaneous measurement of vertical profiles of wind speed and concentration at a point downwind of the emitting area (Smith and Watts 1994a). The mathematical equation for this is shown below (Equation 8).

$$E = \frac{1}{L}\int_0^\infty C(z)u(z)dz \qquad \text{Equation 8}$$

Where:

- *E* is the emission rate;
- *L* is the depth of the source; and
- *C(z)* and *u(z)* are the time averaged concentration and wind speed, respectively at height *z*.

The method has been used successfully by a number of groups. Smith and Watts (1994a) found that with sources of a limited lateral extent and wind directions other than perpendicular the source, the formula upon which the mass balance method is based was no longer appropriate. They also concluded that the greatest limitation of the method is the need for measurement of the full concentration and wind speed profiles. This was important for odour calculations as for odour studies, multiple concentrations cannot be quickly analysed using olfactometry.

2.4.2 Energy balance method

The energy balance method is related to the evaporation prediction of the same name (Smith and Watts 1994a). This method assumes that the emission rate is a

function of the vertical concentration gradient in the horizontal airflow over the source and mathematically, is represented in Equation 9.

$$E = -k\frac{\partial C}{\partial z} \qquad \text{Equation 9}$$

Where

- *E* is the emission rate;
- *C* is the concentration over a height *z*; and
- *k* is a measure of the turbulent mixing in the air flow.

Smith and Watts (1994a) reported that the energy balance method has been used for ammonia and amine emissions from feedlots, however, has not been applied for odour studies. A negative aspect of the method was that it is instrument intensive.

2.4.3 TPS method

The TPS method is a simplification of the mass balance or concentration profile method. Further information on the method can be found in Wilson *et al.* (1983). The method makes use of theoretical wind speed profiles for each of the three main atmospheric stability conditions (stable, neutral and unstable) (Smith and Watts 1994a). Smith and Watts (1994a) concluded that the TPS method was proven and only required minimal data and was limited only by the constraints on source geometry.

Harris *et al.* (1996) described a modification of the TPS by Smith (Smith 1994), which included receptor locations removed some distance from the source and a Gaussian lateral dispersion to cater for sources of limited lateral extent. More recently Zahn *et al.* (2002) and Sommer *et al.* (2004) have used the theoretical profile shape (TPS) method.

Zahn *et al.* (2002) showed that there was significant variation in the calculated emissions over time, but did not compare the data against any direct methods.

In their research, Sommer *et al.* (2004) used the TPS method and another method (bLS method described below) to calculated emissions of gases from feedlot manure stockpiles. They compared this data to the emissions predicted by a flux chamber. They concluded that the flux chamber provided values that were 12-22% of that predicted by the models with the modelled emissions being similar.

2.4.4 Eulerian advection-diffusion

Sarkar and Hobbs (2003) developed a new model based on the Eulerian advection-diffusion equation for a line source. They then extrapolated this to the case of an area source, which is shown as Equation 10.

$$C(x, z_m) = E\int_0^x K' dx \qquad \text{Equation 10}$$

Where:

- C is the concentration at a height of z_m;
- E is the emission rate ($OUm^{-2}s^{-1}$);
- x is the upwind distance; and
- K' is a factor dependant upon wind speed and the shape of the vertical distribution.

Sarkar and Hobbs (2003) used downwind samples taken at a height of 1.5m in the model and compared this to odour emission rates measured using a Lindvall hood (wind tunnel). Overall, they found that the model showed a good agreement with the results from the Lindvall hood.

2.4.5 Backward Lagrangian stochastic model

The backward Lagrangian stochastic model, allows for predictions of the strength of any surface source from one point or line measurements of wind speed and concentration within a downwind plume (Flesch *et al.,* 1995). The model uses the geometry of the surface and the location of a measuring point, to calculate the

trajectories of air parcels upwind from the measuring location (Sommer *et al.,* 2004). The model simulates the ratio of concentration to source flux using Equation 11 and Equation 12 below.

$$E_{bls} = \frac{C_{L,obs}}{(C_L/F)_{sim}} \qquad \text{Equation 11}$$

Where:

- F_{bLS} is the bLS flux;
- $\overline{C_{L,obs}}$ is the concentration in excess of background; and
- *E* is the emission rate (surface flux density) ($OUm^{-2}s^{-1}$).

$$\left(\frac{C_L}{F}\right)_{sim} = \frac{1}{P}\sum_{J=1}^{P}\left(\frac{1}{N}\sum\left|\frac{2}{w_0}\right|\right)\int \qquad \text{Equation 12}$$

Comment [T1]: Check equation

Where:

- *N* is the number of particles released in the simulation;
- w_0 is the vertical touchdown velocities at the source; and
- *P* is the number of specified release points along the line.

2.4.6 Gaussian plume

2.4.6.1 Background

Gaussian plume models are the most commonly used dispersion models. The Gaussian theory assumes that the distribution of the pollutant concentration downwind of a source can be approximated by a normal distribution in the vertical and lateral directions over a sufficiently large averaging time (Harris *et al.,* 1996). The downwind concentration, C(x,y,z), can be calculated from Equation 13 (Smith 1995).

$$C(x,y,z) = \frac{E}{\pi\sigma_y\sigma_z u}\exp\left(-\frac{y^2}{2\sigma_y^2}\right)\exp\left(-\frac{z^2}{2\sigma_z^2}\right) \qquad \text{Equation 13}$$

Where:

- *C* is the concentration averaged over time *t*;

- *x,y* and *z* are the downwind, cross wind and vertical differences measured from the source;
- σ_z and σ_y are the dispersion coefficients representing the crosswind and vertical spread of the plume respectively and which are increasing functions of *x* and *t*; and
- *u* represents the average wind speed.

It is widely acknowledged that Gaussian plume models require a number of assumptions for their use (Harris *et al.,* 1996). These include:

- vertical and horizontal concentration profiles through the plume follow a normal distribution;
- odour obeys the law of conservation of mass; and
- the ground is a barrier to vertical mixing and is a flat surface.

Harris *et al.* (1996) highlighted that wind speed and emission rate are the most important parameter to this type of model followed by atmospheric stability and surface roughness.

2.4.6.2 Stink model

Smith (1995) developed the STINK model specifically to calculate odour emission rates from ground level area sources. During the initial stages of this work it was discovered that adjusting the Monin-Obukhov lengths in the model did not change the model output. As a result of this the model was adjusted to allow for changes in the Monin-Obukhov lengths in the input file. This model uses Gaussian plume theory, assumed vertical wind speed profiles, and, the simultaneous measurement of concentration and wind speed at a point downwind of the source to calculate an emission rate (Smith 1995). Figure 10 shows a schematic of the area source and inputs used for the STINK model. The mathematical basis for the STINK program is given in Equation 14 (Smith 1995).

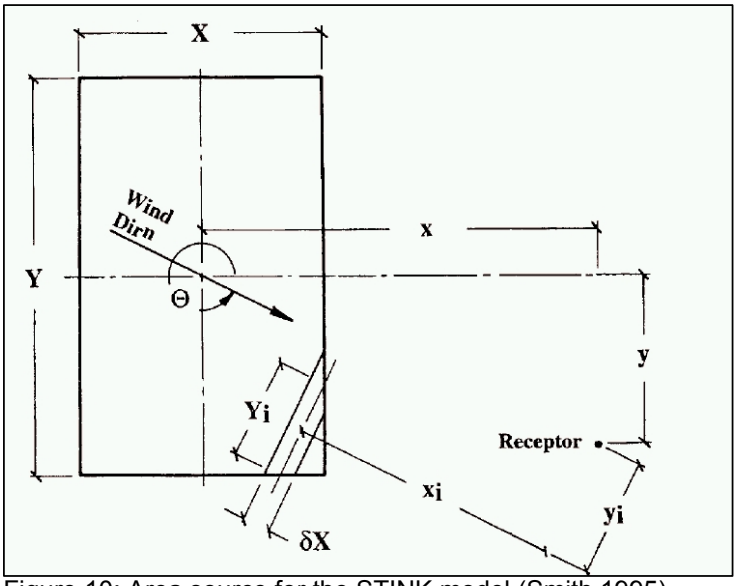

Figure 10: Area source for the STINK model (Smith 1995)

As shown in Figure 10, the source width can be defined as width (X) and the finite length (Y).

$$\Phi(x,y,z)=\sum_{1}^{n}\left\{\frac{1}{\sqrt{2\pi}\sigma_{zi}}\exp\left(-\frac{z^2}{2\sigma_{zi}^2}\right)\left[erf\left(\frac{y+\left(1/2\right)Y}{\sqrt{2}\sigma_{yi}}\right)-erf\left(\frac{y-\left(1/2\right)Y}{\sqrt{2}\sigma_{yi}}\right)\right]\delta X\right\} \qquad \text{Equation 14}$$

Equation 14 uses numerical integration based on the assumption that the area source is comprised of *n* strips of thickness δX. Φ represents the normalised concentration (for emission rate *E* and wind speed *u*) at a receptor located at (*x,y,z*) (Smith 1995). σ_{yi} and σ_{zi} are the dispersion coefficients for each strip. Other requirements of the model include the aerodynamic roughness height of the ground surface z_o.

The Monin-Obukhov length, L, is an alternative measure of stability used in the definition of the vertical wind speed profiles. In the STINK model, it is estimated arbitrarily from the Pasquill stability classes. The length is a function of the friction velocity, reference temperature, density of the air and the vertical heat flux (Harris *et al.,* 1996). The Monin-Obukhov lengths corresponding to each of the Pasquill stability classes were assumed from the table provided by Golder (1972). This

decision was based on the sensitivity analysis (Chapter 3) that showed that the model is fairly insensitive to changes in the Monin-Obukhov length. Table 4 presents the relationship between the Pasquill stability class, the Monin-Obukhov length and the length adopted for use in the model.

Table 4: Pasquill stability and adopted Monin-Obukhov length

Stability Class	Monin-Obukhov length (m)	Adopted length (m)
A		-5
B	$-40 \leq L \leq 12$	-40
C	$-200 \leq L \leq -40$	-200
D	$\|L\| > 1000$	∞
E	$100 \leq L \leq 200$	100
F	$0 \leq L \leq 10$	5

Specific requirements to operate the model include:

- pond width (X m);
- pond length (Y m);
- wind direction (degrees from x axis);
- longitudinal and lateral distances to receptor from centre of the pond (x,y);
- averaging time (hours);
- surface roughness height (m); and
- height for calculation of concentration profile (m).

The model output provides a table of non-dimensional concentrations for increasing sampling heights for different stability classes. An example of the output is shown in Figure 11.

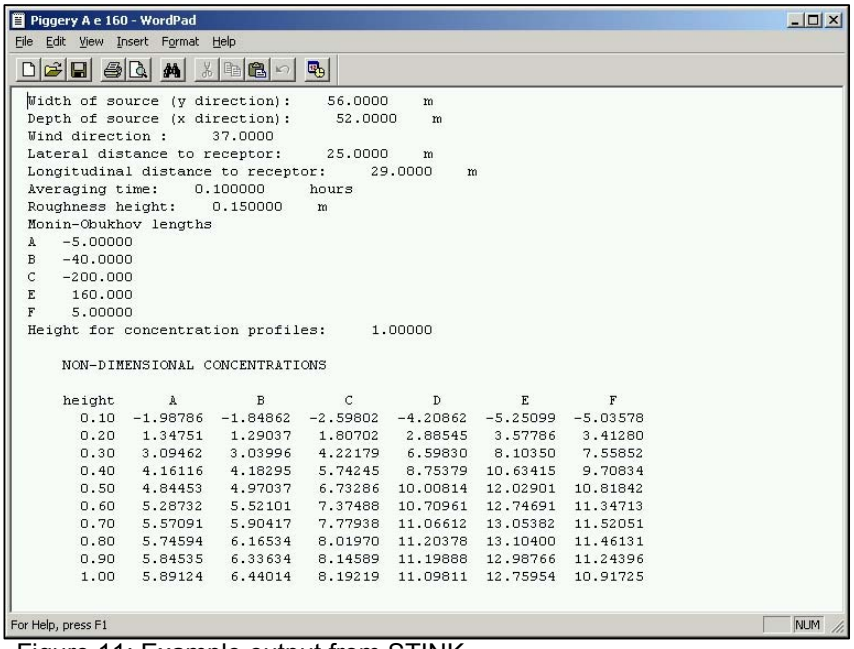

Figure 11: Example output from STINK

The model can allow for different sampling heights, however, for this work all samples were taken at a height of one metre. The non-dimensional concentration for the emission rate calculations is selected based on the stability class at the time of sampling and the height at which it was taken. This data is easily obtained from the output shown above. The emission rate is then calculated using Equation 15 (Smith 1995).

$$E_a = \frac{C(z)\hat{u}}{\psi(z)} \qquad \text{Equation 15}$$

Where:

- E_a is the calculated emission rate ($OUm^{-2}s^{-1}$);
- $C(z)$ is the measured odour concentration (OU/m^3) at height *z*;
- $\hat{u}$ is the wind velocity measured on site (m/s); and
- $\Psi(z)$ is the non-dimensional concentration at the sampling height.

A sensitivity analysis was undertaken by Smith (1993) that identified the following variables which affect odour concentration downwind from an areal source:

- odour emission rate;
- wind speed and atmospheric stability;
- dispersion coefficients;
- averaging time;
- ground surface roughness height;
- mean wind direction;
- standard deviation of wind direction; and
- strip width.

Harris *et al.* (1996) identified wind speed and emission rate as the most important parameters followed by atmospheric stability and surface roughness. Wind speed was found to have two effects on odour concentration. The normalised concentration (Equation 15) is inversely related to wind speed and that wind speed at a height of 10m is a factor in the selection of the atmospheric stability classification. Smith (1993) concluded that there was nothing to suggest that the results would be significantly different for concentrations close to an area source.

He also found that varying the stability class used by one interval (which was the largest error likely to occur when calculating the stability class) an increase in concentration of between 40% and 90% was observed. Varying the stability class is equivalent to varying the dispersion coefficients simultaneously (Smith 1993). Varying of the coefficients individually, measures the effect of uncertainty in the schemes used to estimate the parameters, which involves a redistribution of the odour through the normal distribution (Smith 1993).

Smith (1993) found that concentration was only moderately sensitive to the roughness height, Z_o but due to the fact that roughness height is difficult to estimate it is an important factor in dispersion modelling. The averaging time *t* was found to be unimportant as it can be measured reasonable accurately.

Smith (1993) summarised that for concentrations at 1m in height and close to an areal source (<1000m), the parameters in order of decreasing importance would appear to be:

1. wind velocity;
2. emission rate;
3. stability class;
4. roughness height;
5. vertical dispersion coefficient (σ_z);
6. mean wind direction (θ);
7. averaging time;
8. horizontal dispersion coefficient (σ_y); and
9. the standard deviation of the wind direction (σ_θ).

2.5 Verification of dispersion models

A number of studies have examined the accuracy of predictions from dispersion models. Whilst there is no doubt that the models work well for relatively inert materials such as certain particulate matter and gases, their use for odour is based on a number of assumptions as detailed above.

As a result of the uncertainties, studies into the use of modelling to determine the impact of emissions from area sources have been undertaken. These include those of Koppolu *et al.* (2002), Sarkar and Hobbs (2003), Edgar *et al.* (2002), Guo *et al.* (2001), Sommer *et al.* (2004) and Smith and Kelly (1996).

The works listed above, can be broken down into two categories. These being the use of models which predict the impact from the use of emissions data (regulatory application) or models that are used to calculate emissions data from downwind concentrations (back calculation models).

2.5.1 Comparison of STINK and TPS methods

The Lagrangian TPS method of Smith (1994) and the STINK method of Smith (1995) were compared by Harris *et al.* (1996). They found that at 50 to 200 metres from a source, both models gave almost identical concentrations at all heights for neutral conditions. However under very stable and very unstable conditions the correlation between the results from the two models was poor. They hypothesised that this could be due to the different descriptions of stability used in the two models.

2.5.2 Verification of regulatory models

When modelling odour, the underlying problem with back calculation from area sources is the uncertainty inherent in olfactometry (Smith and Kelly 1995). Guo *et al.* (2001) and Edgar *et al.* (2002) undertook verification studies on the effectiveness of Gaussian models (INPUFF and Ausplume respectively) for assessing odour dispersion. Generally these models are used to predict the number of hours per year which the odour exceeds a certain concentration (impact criteria).

The difference between the two approaches was that Guo *et al.* (2001) used nearby residents to assess odour over a period of time and then correlated their responses to those predicted by the model. They did this by comparing the resident's responses with n-butanol reference concentrations. In their study, Edgar *et al.* (2002) used downwind samples analysed using olfactometry to compare measured odour concentrations with those predicted by the model.

Guo *et al.* (2001) found that the INPUFF-2 model satisfactorily predicted faint odours (low odour intensity) from sources up to 3.2km away under stable to slightly unstable conditions. However, it was also found that the model underestimated the dispersion of moderate to strong or very strong odours and under predicted odour levels during neutral or unstable weather as compared to the field measured data.

They also highlighted that under calm conditions (wind speeds less than 0.5 m/s) the model was not capable of predicting odour dispersion properly. Gaussian dispersion

cannot handle low wind speed events under stable conditions (NZMfe 2004). The finding of Guo *et al.* (2001) using the Inpuff-2 model was unexpected as puff models are generally assumed to be capable of handling dispersion better than the standard Gaussian models.

The USEPA recommends replacing low wind speed values with a default value of 1m/s. This is a function of the stall speed of most cup anemometers. In Australia this approach is followed however, values less than 0.5 m/s are replaced by a value of 0.5 m/s. This is an important aspect when using Gaussian models, as the models do not provide realistic concentration estimation at wind speeds less than 0.5m/s. With the more modern models such as Calpuff, much lower wind speeds can be modelled accurately.

In their work Edgar *et al.* (2002) found limitations with the Gaussian Ausplume model. As discussed briefly above, the limitations of Gaussian models are well known and include:

- that the models assume that pollutant material is transported in a straight line instantly to receptors (NZMfe 2004);
- the model breaks down at wind speeds less than 0.5 m/s (Guo *et al.,* 2001; PAE 2003b);
- in moderate terrain areas they will typically overestimate terrain impingement effects during stable conditions (NZMfe 2004);
- they have no memory of the previous hours emissions (NZMfe 2004); and
- the models assume that the atmosphere is uniform across the modelling domain (NZMfe 2004).

The limitations of Gaussian models play an important role in the findings. The slight slope in terrain listed by Edgar *et al.* (2002) could be the cause of the differences as the terrain would play an important effect due to night-time valley drainage.

They also found that the model under-predicted odour concentrations at the centreline of the odour plume and over–predicted the spread of the plume at distances within approximately 1000m of the odour source. A significant finding of this work was that the odour in the plume from the piggery did not appear to be conserved, which is an assumption on which Gaussian models are based (Edgar *et al.,* 2002). For in depth information on the differences between the various models see Ormerod (2001), NZMFE (2004) or Diosey *et al.* (2002).

2.5.3 General back calculation studies

A number of groups have back calculated gaseous emissions from area sources using non-regulatory models. These include Koppolu *et al.* (2002), Smith and Kelly (1995; 1996) and Sarkar and Hobbs (2003)..

Sarkar and Hobbs (2003) compared a Lindvall hood and a Eulerian advection-diffusion flux footprint method to estimate odour fluxes from a sold waste landfill site. The Eulerian based model was shown by Walcek (2003) to be superior for modelling the effects of wind shear, an effect that is ignored by Lagrangian puff models.

Sarkar and Hobbs (2003) found that their back calculation method (bLS model) showed promise when compared to emissions calculated by a Lindvall hoods. Their study was limited by resource constraints, which meant there was a limited dataset. They concluded that the use of a back calculation method provided the major advantage of analysing emissions when the surface was under natural meteorological conditions.

Koppolu *et al.* (2002) used a modified version of the STINK model which included Draxler's approach (Draxler 1976) to back calculate volatile fatty acid emissions from an area source. Their aim was to utilise back-calculation methodology to assess emission rates of specific gases for near source dispersion. They used five receptors ranging from 2 m from the source to 20 m from the source. At each of these receptors, SME fibres were used to collect samples and later analysed by GCMS to

determine gas concentrations. Three of the receptors were along a centreline directly downwind of the source with an additional two located on each side of the centreline in an attempt to assess the edges of the plume. The source consisted of four children's swimming pools placed adjacent to each other in a square shape. The base was approximately 2 m x 2 m.

They found that the volatile fatty acid concentrations predicted at the receptors located off the centreline of the plume were unrealistic in some situations partly attributable to plume fluctuations (Koppolu *et al.,* 2002). They concluded that the model performed well under the given conditions but it did not produce results similar to the theoretical 1:1 relationship (Koppolu *et al.,* 2002).

Sommer *et al.* (2004) conducted an in depth emissions study of emissions from feedlot manure. They used two back calculation methods and a flux chamber type device. Whilst the two back calculation methods compared reasonably well, the flux chamber emission rate was only 12-22% of that calculated using the models (Sommer *et al.,* 2004).

In general back calculation techniques have been applied successfully, i.e. they have calculated an emission rate. However the ability to define what is the "real" emission rate compared to those derived from wind tunnels and flux chambers hampers defining the most appropriate method for area sources.

2.6 Spatial variability of odour emissions

Currently there is little known about the spatial dynamics of emissions from piggery lagoons. While not stated explicitly, publications regarding odour and VOC sampling implies that the surface liquor of anaerobic ponds is well mixed and spatially homogenous (Hudson *et al.,* 2004). This indicates that the emissions from piggery lagoons do not vary spatially. Jiang (2002), Heber *et al.* (2000) and Bicudo *et al.* (2002) have all undertaken measurement of odour emissions from anaerobic

lagoons. Unfortunately Jiang's (2002) work has not detailed any emission rates or the number or location of samples taken. Heber *et al.* (2000) and Bicudo *et al.* (2002) measured odour emissions at a single point and did not take samples over the ponds surface.

Previous research has eluded to the fact that difference seen between back calculation and wind tunnel methods may be due to the inability of single samples taken using a wind tunnel to be representative of the true average emission rate of an entire area source (Smith and Kelly 1996).

A study by Melbourne Water at the Werribee Sewage Treatment Plant showed large variations in emission rates over the surface of their anaerobic lagoons (0.05 – 7.3 OU $m^3/m^2.s$) (Gulovsen *et al.,* 1992). More recent studies at sewage treatment plants in Western Australia have shown significant spatial variation in emissions in primary sedimentation tanks and activated sludge tanks (Gwynne *et al.,* 2002). Overall, research undertaken to date has not detailed enough information to ascertain the existence of spatial variability or to draw further conclusions.

Chapter 3 STINK sensitivity analysis

3.1 Introduction

A model can only be used to its full potential if good quality data is available to the user. Thus, prior to using any model, the importance of each input parameter should be assessed to determine the influence of errors and uncertainties in the input parameters on the output.

The objective of this sensitivity analysis was to enable future users of the model to make informed decisions regarding the importance of parameter selection and input with a view to determining an accurate estimate of emission for samples taken close to an area source.

The sensitivity analysis was undertaken using the data derived from this project and the variables identified by Smith (1993). These included roughness height, atmospheric stability, Monin-Obukhov length, averaging time, variation in source dimensions and effect of input wind direction. When undertaking the sensitivity analysis for this project, all variables with the exception of the variable in question were kept constant. The variable in question was then modelled using a range of values, to assess possible impacts. Piggery C was selected at random to undertake the sensitivity analysis upon.

3.2 Results

The results from the sensitivity analysis on the variables detailed above are given in Figure 12 to Figure 20.

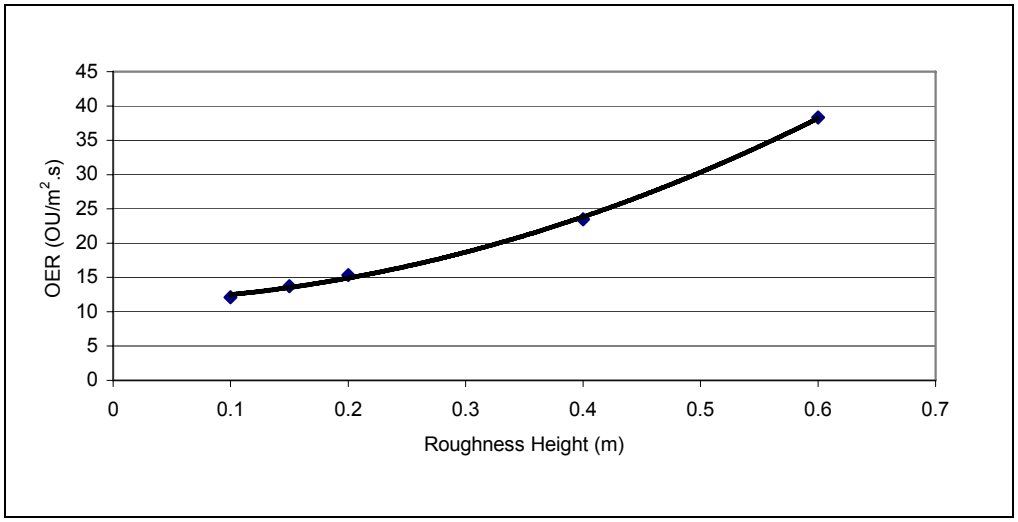

Figure 12: Effect of varying roughness height

Figure 12 shows that increasing roughness height, can result in increased prediction of odour emission rate.

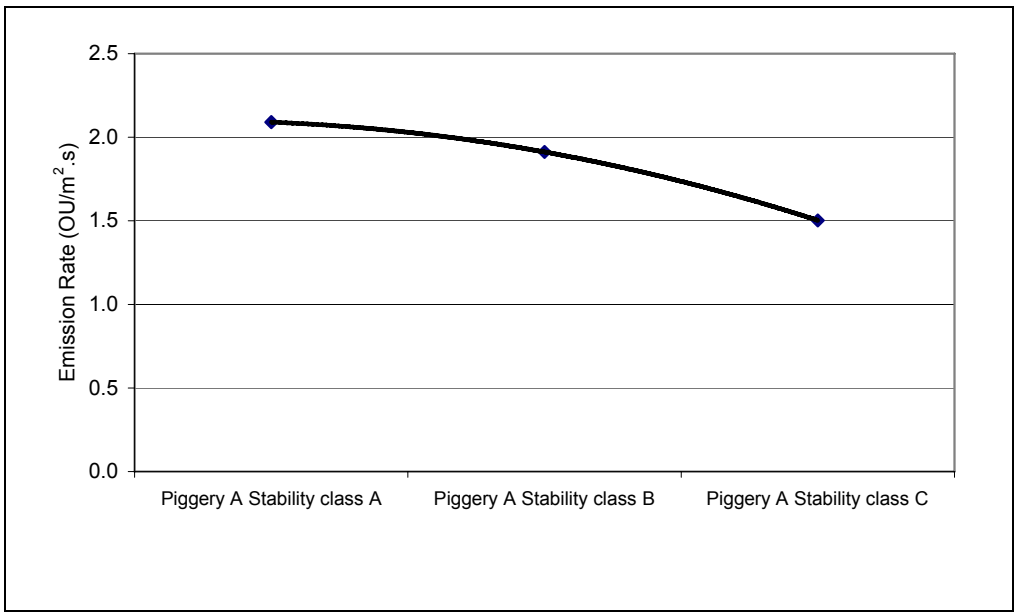

Figure 13: Result of Variation in Stability Class

Figure 13 shows that varying stability class by one, can result in moderate changes in the predicted odour emission rate.

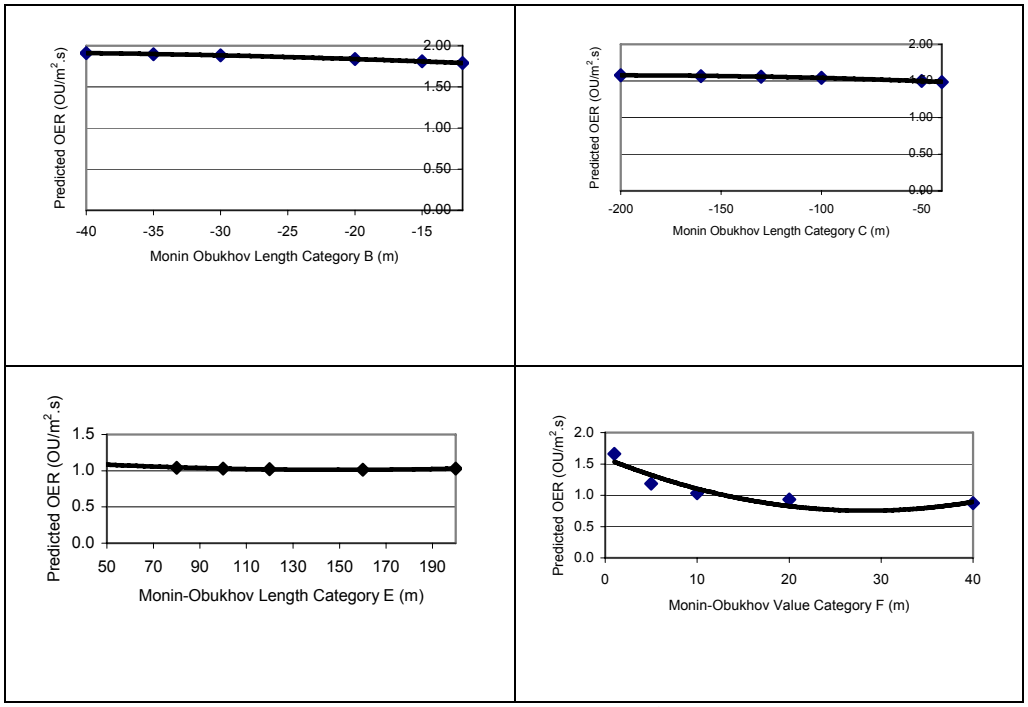

Figure 14: Variation associated with changes in Monin-Obukhov length

It can be seen in Figure 14 that under most classes, the length has little influence on the predicted OER. However, for stability class F it can have a large influence on the predicted OER as the length (L) approaches zero.

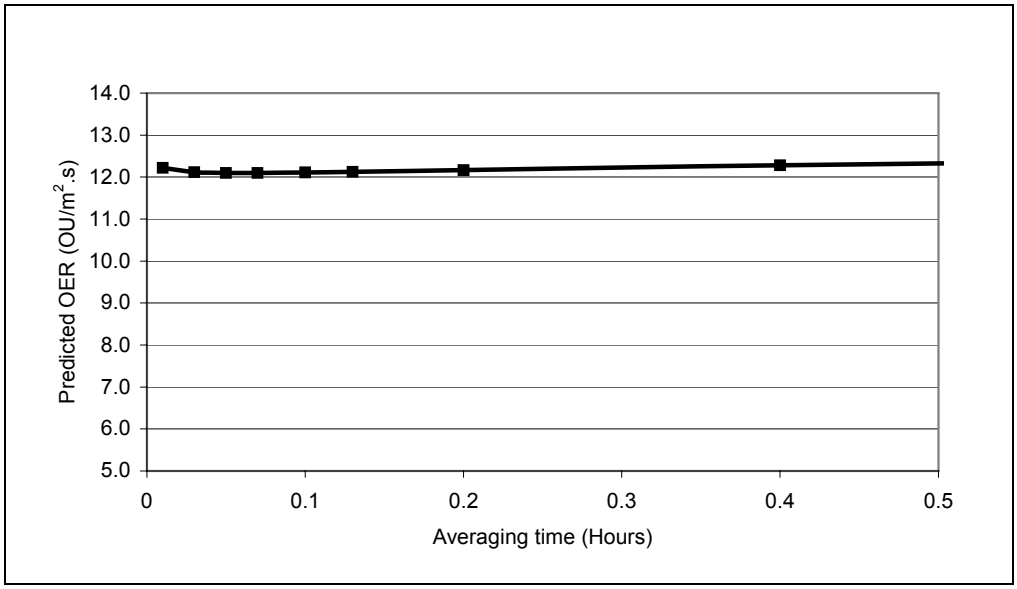

Figure 15: Effect of variation in averaging time for the STINK model

It can be seen in Figure 15 that averaging time has very little effect on the output of the model.

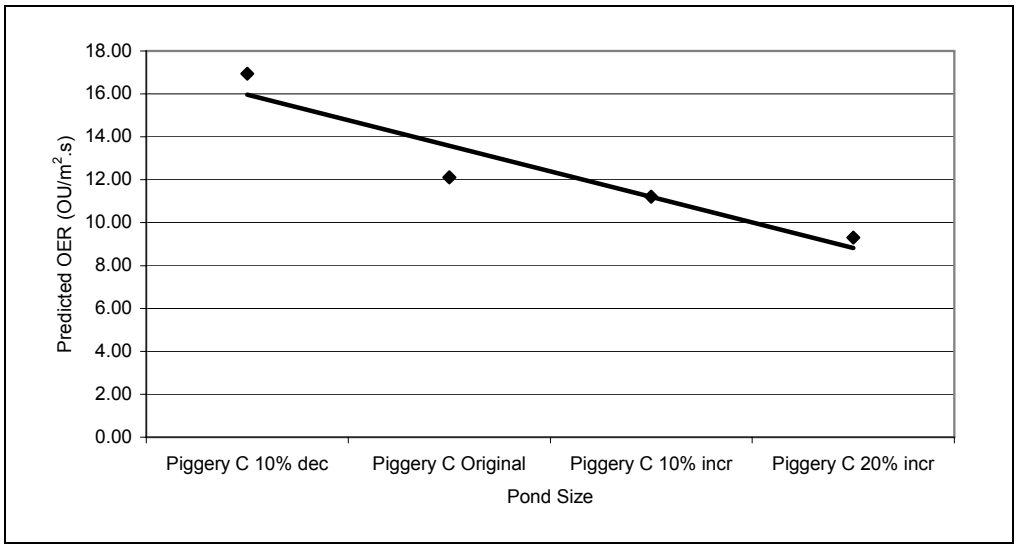

Figure 16: Effect of variation in dimensions for Piggery C

During this process a discrepancy was found between this work and the sensitivity analysis of Harris *et al.* (1996). Further investigation showed that Piggery C had a

long and narrow pond and was the only pond studied with this design. Therefore, it was decided to investigate the impact of pond dimensions on the pond at Piggery E which was the largest pond studied and had a square pond surface (147 x 116m).

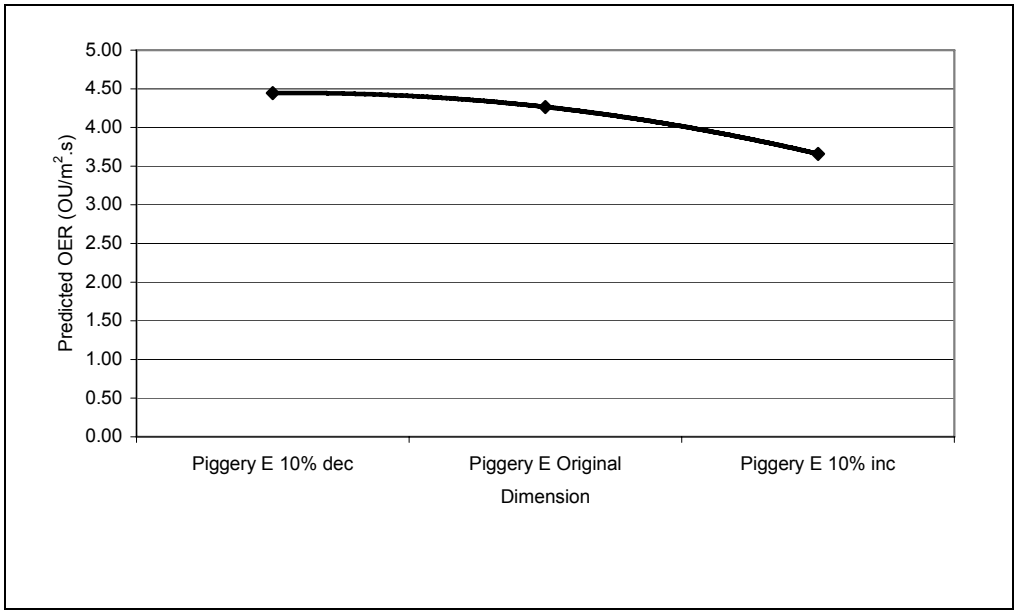

Figure 17: Effect of variation in dimensions for Piggery E

Figure 17 shows that a 10% change in dimensions changes the calculated emission rate by approximately 20%. The model uses x and y inputs to define source dimension and as such can only handle square/rectangular ponds. To overcome this obstacle the modeller generally defines the length and width of the source as the model's X and Y inputs. Then the wind direction is altered by the angle in which the pond was different from the North axis. This in effect "tells" the model that the pond was aligned as per the model design. To assess variation in emissions in the case of a modeller not adjusting the wind direction, a comparison was made of the use of easily defined width and length values and the more accurate area calculated from the GPS survey. The width and the depth of the pond used in the research were found to be 40m x 130m. To achieve an area similar to the real area, values of 36.6 and 118.95 were substituted.

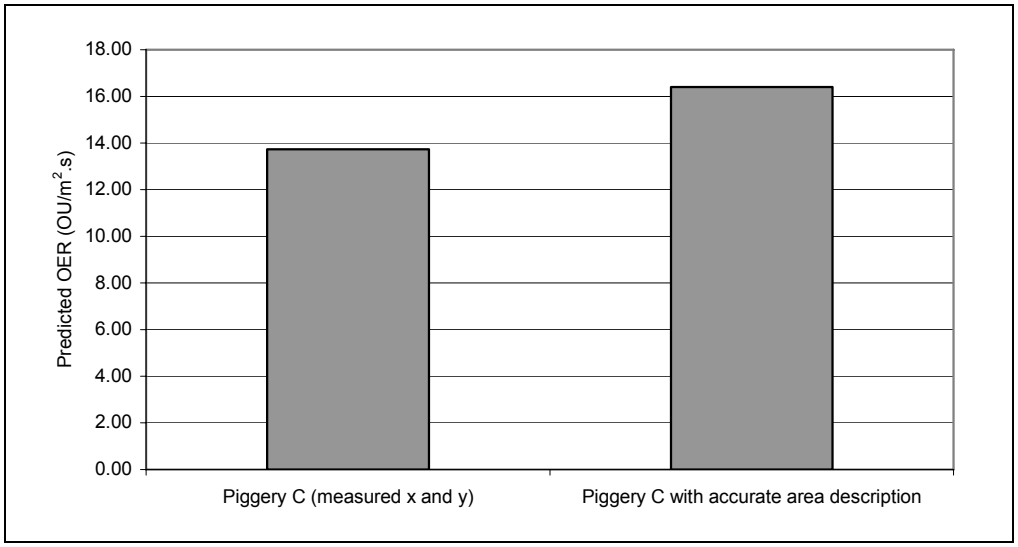

Figure 18: Difference in Emissions with variation in Area

Figure 18 shows that changes in the dimension can have an impact (≈15%) on the predicted OER for sample sites close to a source. To assess the variation in wind direction on the model output, values of 5, 7, 9, 11 and 15 degrees were modelled.

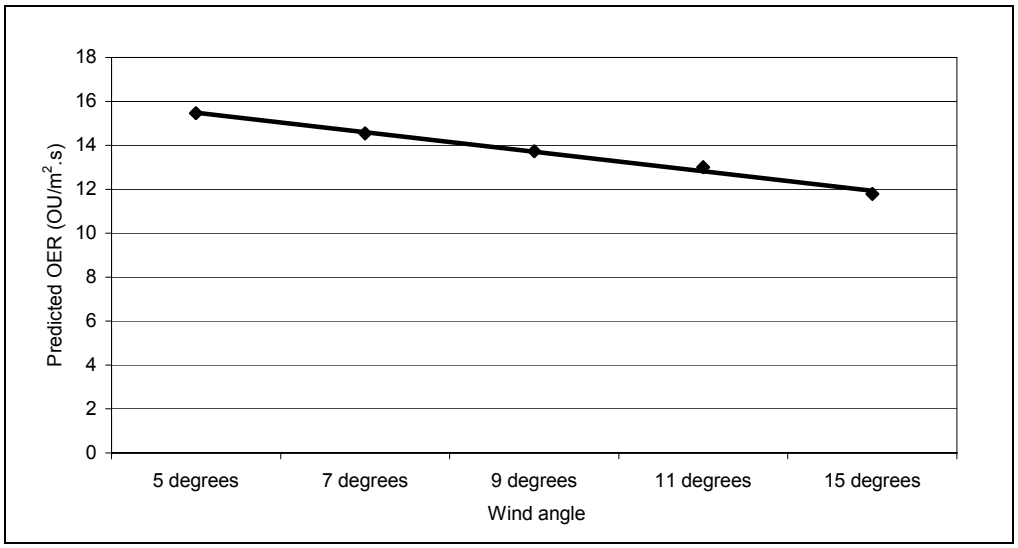

Figure 19: Effect of change in wind angle

Figure 19 shows that the effect of change in wind angle is relatively constant over the range tested (5-15°) and has a smaller effect on the predicted OERs than other factors.

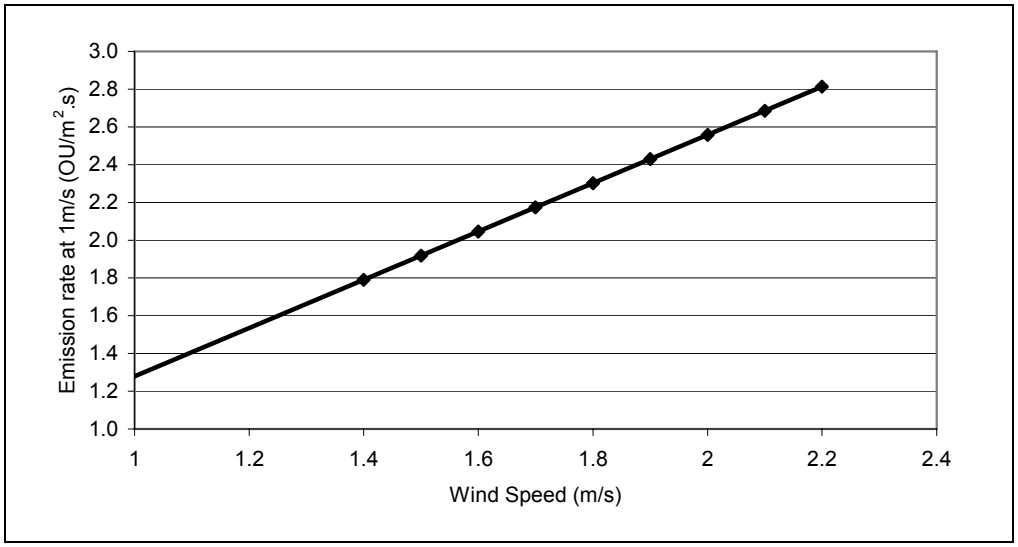

Figure 20: Effect of change in wind speed

As expected, Figure 20 shows that increasing wind speed increased the predicted OER.

3.3 Discussion

During any modelling exercise, the effect of model inputs (i.e. sensitivity analysis) should known. This is to 1) gain an insight into the models abilities and 2) to examine areas in which input data could significantly affect predictions.

Smith (1993) identified wind speed, atmospheric stability, dispersion coefficients, averaging time, surface roughness, mean wind direction and the strip width used in the STINK model as parameters that could affect the model predictions. Of these, wind speed and odour emission rate were identified as most important, followed by atmospheric stability and surface roughness. He also concluded that the vertical dispersion coefficient and wind direction was of moderate importance (Harris *et al.,* 1996; Smith 1993). As mentioned previously the model was updated to allow for

adjustment of the Monin-Obukhov lengths. As a result of this the model was adjusted to allow this. Thus to confirm the findings of earlier sensitivity analyses, the analysis was again based on the following variables as identified by Smith (1993):

- Roughness height;
- Atmospheric stability;
- Monin-Obukhov length;
- Averaging time;
- Variation in source dimensions; and
- Effect of input wind direction.

3.3.1 Effect of input wind angle and wind speed

Harris *et al.* (1996) concluded that the input values for wind direction for STINK was unimportant but wind speed was a critical parameter. This was highlighted in the data shown in Figure 20. The effect of wind speed on the modelling process is twofold (Smith 1993):

- The downwind concentration is inversely related to the average wind speed; and
- That the wind speed at 10 m is a factor in the selection of stability classes used in the model thus affecting the models outputs.

As shown in Figure 20, changing wind speed (irrespective of stability class) has a large impact upon the predicted odour emission rate. As wind speed and stability class are linked, errors associated with measuring wind speed can be transferred through to the calculation of the stability classes. The accuracy of the sensors used during this project was ±1.5% for the wind speed and ±3 degrees (≈±1% of total) for wind direction. Thus it is unlikely that small errors in the measurement of wind speed would cause significant changes in the predicted odour emission rate. Errors in direction either via measurement on aligning the wind vane with north would be more likely to impact upon the results. Smith (1993) found for an odour sample collected

close to an areal source, the effect of variation in measured wind direction was negligible. For the sensitivity event, the wind direction was 9° from the x-axis. To assess the variation in wind direction on the model output, values of 5°, 7°, 9°, 11° and 15° were modelled with the results shown in Figure 17. The results showed that a change of approximately 4° results in a change in the predicted odour emission rate of about 9% thus indicating that small errors in wind direction would have a minimal effect on models output when compared to other factors such as wind speed or stability class.

3.3.2 Surface roughness

Roughness height is a standard input into dispersion models and is used to represent the presence of features in the surrounding landscape. The variation likely in this parameter makes it extremely important in dispersion modelling (Smith 1993) for distances away from an area source (Harris *et al.,* 1996). The USEPA (2001) recommends a value of 0.1 metres for low crops and occasional large obstacles and a value of 0.25 metres for high crops with scattered obstacles. Values of between 0.003m to 0.1m for grass and 0.04 to 0.2m for agricultural crops have been previously published (Smith 1993).

To assess roughness height, Piggery C was selected at random. Although it was a long narrow pond, this design should not have unduly affected the model's output as the exercise was representative of a real world modelling exercise. Harris *et al.* (1996) showed for the STINK model, the non dimensional concentration (raw output from the model not the calculated emission rate), was relatively insensitive to small changes or errors in the measurement or estimates of aerodynamic roughness. The roughness heights for the sample sites were estimated to be between 0.1 and 0.2m. These estimates were loosely based upon the 1/10th rule of thumb, in which the roughness height for a site is the height of any obstructions divided by 10. Because of these estimates, surface roughness was increased in the model from 0.1 to 0.6 m

to assess changes in the predicted odour emission rate. As shown in Figure 12, increasing surface roughness from 0.1 to 0.15m increased the emission rate by approximately 10% and increasing the roughness height from 0.15 to 0.2 metres increased the emission rate by a further 10%. From changing the roughness height two conclusions have been drawn. Firstly, for a value between 0.1 and 0 .2 metres, the overall impact of changing surface roughness does not appear to be as great for roughness heights above 0.2 metres and secondly, relatively large changes in surface roughness (i.e. doubling from 0.1 to 0.2) has little effect on the predicted emission rate.

There are prescriptive methods available to describe roughness height, however, due to the difficulties in fully describing a limited upwind fetch at a piggery, a value of 0.15m was deemed appropriate for modelling, as it provided a realistic estimation of emission rate compared to the wind tunnel and additionally, limited additional variation being included.

3.3.3 Atmospheric stability

Stability classes describe the tendency of the atmosphere to enhance or restrict horizontal and vertical movement of an odour plume.

As described previously, atmospheric stability was calculated using the sigma-A method. To assess the impact of the stability class on the model output, the stability class at the sampling time was varied by one interval to mimic the sensitivity analysis of Smith (1993). For the analysis, Piggery B was selected. The results from this assessment were shown Figure 13. The results demonstrated that incorrect assignment of stability class by one class could result in a change in predicted odour emission rate by +/- 30%. Thus it is concluded that the model is relatively sensitive to stability class selection. As discussed below, the Sigma-A methods prediction of stability class differs from Turners and SRDT methods under stable conditions. Thus

errors in stability class determination may occur if a) the meteorological data is erroneous or b) sampling is restricted to very early morning or night time.

3.3.4 Monin-Obukhov length

Atmospheric stability is also represented in the model by the Monin-Obukhov length. The Monin-Obukhov lengths corresponding to each of the Pasquill stability classes were based on the table provided by Golder (1972). Assumed Monin-Obukhov length values may not be identical to those that would have been measured during sampling making it important to assess the impact variable selection may have on model output.

During the sensitivity analysis it was noted that varying the value of Monin-Obukhov lengths did not significantly influence modelled prediction. This was due to the Monin-Obukhov lengths being “hard wired” into the program and not responding to user input. The model was subsequently updated by Professor Rod Smith of the University of Southern Queensland to allow users to vary the Monin-Obukhov lengths. A series of values were then assessed for their effect on model output. The result of variation in the lengths under different stability classes was previously shown in Figure 14.

It was found that only stability class F showed a significant variation in odour emission rate. No samples were collected during this project under these stability conditions, so the use of assumed Monin-Obukhov lengths is appropriate. This does indicate that if the model were to be used under stable conditions, site-specific Monin-Obukhov lengths would be beneficial. Under other conditions selection of values of this parameter appear to play a minor role in the overall function of the model.

3.3.5 Averaging time

Averaging time refers to the period over which the odour sample was collected or alternatively over which a model predicts a concentration. Based on the previous

use of the model by Smith and Kelly (1995), a 6-minute averaging period was selected both for modelling and sample collection. To gain an insight into the impact of averaging time selection on model output the averaging time was varied from 0.01 to 2 hours. The results of this assessment were shown in Figure 15.

Figure 15 shows that variation in averaging time had a minor effect on the output of the model. However, this did not take into account temporal variation of odour emissions during the modelling or sampling process. From the limited modelling, it is concluded that the time over which a sample was taken for this model, is relatively unimportant.

3.3.6 Variation in pond dimensions

Harris *et al.* (1996) discovered that pond size had negligible impact on the modelled odour emission rate. Initially the impact of pond size was assessed using a trial pond (Piggery C) with a 10% decrease and a 10% and 20% increase in pond dimensions. The results of the assessment (Figure 16) indicated that the dimensions of the pond were important when determining odour emission rates using the STINK model. This is contrary to the findings of Harris *et al.* (1996) and Smith (1993) who concluded that the dimensions were not an important factor when determining odour emission rates. Further examination revealed that Piggery C was long and narrow (133 x 35m, Figure 21)). This would have impacted on the predicted odour emission rate, as the pond had a much smaller overall surface area and was irregular in shape. Since the adoption of the Rational Design Standard (Barth 1985) in Queensland, very few ponds have been built which are not rectangular or square (with a length to width ratio that is no more than 2:1).

It was decided to investigate the impact of pond dimensions on the model output for Piggery E (147 x 116m) that was deemed more representative of the current design rationale for ponds.

The data presented in Figure 16 showed that a 10% decrease in pond dimensions results in a 20% increase in predicted emissions for Piggery C. The predicted emission rate for Piggery E increased by 4% (Figure 17). Whilst a 10% increase in pond dimensions caused a 20% decrease in predicted emission rate for Piggery C and a 15% decrease for Piggery E.

Although this data indicates that pond dimensions are not as important as other factors in determining odour emission rates for regularly shaped ponds, pond shape still should be viewed as an important factor when modelling.

IN the STINK model, the dimensional input is somewhat simplistic when compared to other available models, such as Ausplume and Windtrax, where irregular shapes can be modelled. The model uses an X and Y input to describe the emitting source and as such can only handle square/rectangular ponds. Generally, the user of the STINK model defines the length and width of the source as the model's X and Y inputs. Then the wind direction is altered by the angle in which the pond was different from the North axis. This in effect "tells" the model that the pond was aligned as per the model design.

To assess what would happen if the wind direction were not altered, a comparison was undertaken on pond dimensions and their effect on predicted odour emission rates. The width and the depth of the pond were 35m x 133m (area of 4655 m^2) based upon an initial survey. A GPS survey indicated that the actual dimensions were 36.6 and 118.95m (area of 4353 m^2). These data were input into the model and the results of this assessment were shown in Figure 18.

The results showed that use of better-defined dimensions resulted in the calculation of an emission rate that was approximately 20% greater than that derived using less accurate pond dimensions. Thus, it can be concluded that for a regularly shaped pond, small discrepancies in the measurement of the length and width inputs have some effect on the output of the model. For irregularly shaped ponds, pond

dimensions may have a significant impact on calculated odour emission rate if the pond alignment is not adjusted as described above.

3.4 Conclusion

The sensitivity analysis showed that a number of inputs are important for the STINK model when determining emissions from close to an area source. The variables in order of importance for samples taken close to a source were found to be:

1. Wind speed;
2. Atmospheric stability;
3. Pond size and alignment;
4. Surface roughness;
5. Wind angle/direction;
6. Averaging time; and
7. Monin-Obukhov length (except under stable night time conditions as L approaches zero).

Chapter 4 Methodology

4.1 Olfactometry

In 2000, the Queensland Department of Primary Industries and Fisheries upgraded its olfactometer to automate dilution indexing and to incorporate other changes as to conform to the Australian Standard for Dynamic Olfactometry (AS4323.3) (Standards Australia 2001). Elements of this upgrade have been discussed in Zeller *et al.* (2002) and Nicholas *et al.* (1999).

The olfactometer can undertake dilutions from 2^1 to 2^{15} (1 to 32768 dilutions), thus, when an appropriate amount of sample is available the lower detection limit is 6 OU. For a sample size of around 100 litres, the minimum concentration measurable is around 10 OU.

The olfactometer is a three way forced choice dynamic olfactometer. Three way refers to each panellist having three ports to sniff from. The panellists are all presented with an odorous sample, which has been diluted to a predetermined concentration. The panellists are asked to respond whether they were certain, uncertain or were guessing which port the odour is being emitted from. If, during the first round, a panellist responds certain and correct (i.e. they could detect the odour with certainty) the odour is further diluted and the odour is presented to the panel at the lower concentration (higher dilution). This process occurs until the odour cannot be determined correctly with certainty by the panel members. The concentration is then doubled and presented again to the panel. This occurs until all of the panel members have responded certain and correct twice to the odour (Standards Australia 2001). This is defined as a round. A Z_{ITE} or individual threshold estimate is then calculated for each panellist for that round. Two more rounds are undertaken using

the above-mentioned process. This process has previously been described in section 2.1.3.2.

4.2 Meteorological data collection

4.2.1 Weather station

Site-specific meteorology was measured using custom built weather stations. The stations measure wind speed, direction, rainfall and incoming solar radiation all at a height of 2m.

4.2.2 Stability class calculation

Stability classes were calculated using the Sigma-A method (USEPA 2000a). The method is turbulence based that uses the standard deviation of the wind direction in combination with the scalar mean wind speed (USEPA 2000a). There are a number of methods available to determine stability class and recent work by Mohan and Siddiqui (1998) and Tripp *et al.* (2004) has shown that there are some inconsistencies in stability classes determined using the USEPA methods. However, given the limited data required for the Sigma-A method, it was deemed appropriate for this work.

4.3 Sample site selection

4.3.1 Pond selection

The ponds used in this project had to meet four specific criteria. The criteria were:

- the pond had a known operating history;
- a volatile solids loading rate could be easily determined;
- there was only one primary pond; and
- there were only one or two discharge points into the pond.

The selected ponds gave a representation of a number of different volatile solids loading rates. Sampling sites on each pond were selected beyond the batter of the

pond to ensure that there was a uniform depth of effluent below each sampling point. To create a sampling grid, the surface area of the pond was broken up into sections with equidistant spacing within the grids. Samples were taken in the centre of each grid. Table 5 shows the characteristics of the ponds studied for the project.

Table 5: Pond Characteristics

Piggery	Pond length (m)	Pond width (m)	Depth[2] (m)	Pond Volume (ML)	Surface Area (m^2)
A	56	52	6.0	6.0	2946
B	64	54	3.2	10.8	2977
C	133	35	4.0	8.7	4600
D	110	76	5.5	27	8517
E	146	116	7.0	73	16882
F	45	34	4.3	4.3	1154

4.3.2 Sample grid selection

The sampling grids were selected as to be proportional to the pond size based on the data in Table 5 and are shown in Figure 21. A minimum of six samples were taken on the smallest pond with up to nine samples being taken on the larger ponds.

[2] Measured depth to base of pond

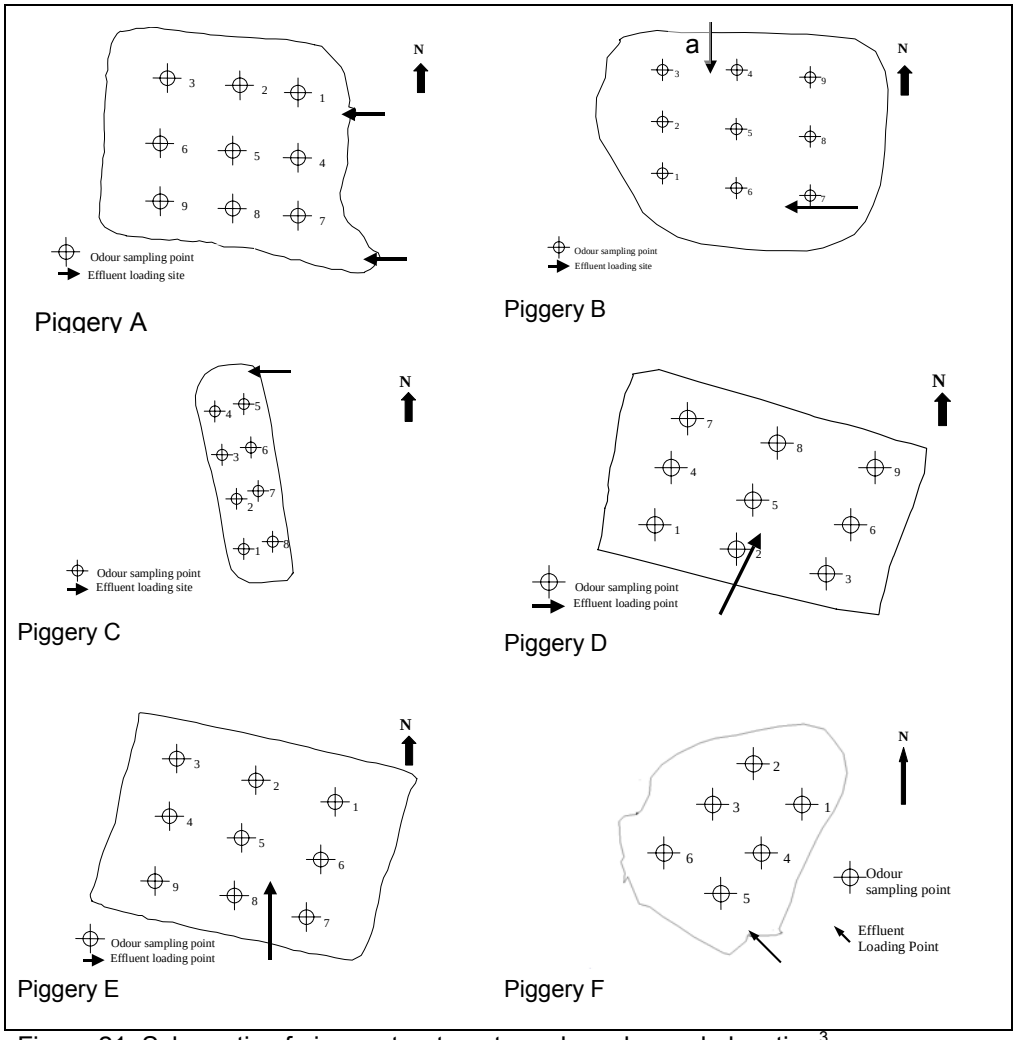

Figure 21: Schematic of piggery treatment ponds and sample location[3]

4.4 Emission rate determination

4.4.1 Sampling

Samples were taken during the summer of 2000-2001, winter 2001 and the summer of 2001-2002. Each piggery was sampled once during each period. Additionally, Piggery B was sampled again, at the end of the sampling period to evaluate any

[3] Note: Sites are not drawn to scale.

variability during the sampling period. Sampling was undertaken on Tuesday and Thursday in one week for each pond to assess variability between days.

The average odour emission rate for the sampling day was the averaged odour emission rates for all sample points for a pond.

4.4.2 Stink modelling

Where possible, a minimum of two odour samples were taken downwind of each pond studied for each sample day. Samples were taken at a height of 1 metre, approximately 1 metre from the edge of the pond. Wind speed was determined using a Thermo Systems Incorporated (TSI) Model 8355 hot wire anemometer with downwind samples only being taken when wind was at or above a velocity of 1 m/s. This wind speed was not used for the modelling process but to determine optimum sampling times. Sampling was not undertaken if another source (i.e. secondary pond or sheds) were upwind of the downwind sample site.

The velocity used for the modelling was sourced from the on site meteorological station. A surface roughness height of 0.15 m was adopted.

The required inputs were placed into the model and an emission rate was calculated. To enable the emission rate to be scaled to a standard wind speed of 1 m/s inside of the wind tunnel the wind profile for the site was calculated using the power law (Equation 7).

The calculated odour emission rate was then scaled to a standard wind speed of 1 m/s inside of the tunnel using Equation 6, the factor of 0.5 (Bliss *et al.,* 1995; Pollock 1997) and the velocity obtained using Equation 7.

4.4.3 Calculation of on pond odour emission rates

Odour samples were collected using a University of New South Wales wind tunnel operated according to standard protocol (i.e. Bliss *et al.* (1995) and Jiang *et al.* (1995)). When measuring the cross sectional velocity profile at the tunnel, it was

found that the exit point exhibited a skewed velocity profile, not a normal distribution. To even the flow profile a 4” PVC extension was used to increase the length of the section, promote laminar flow and thus flatten the velocity profile. As this did not sufficiently flatten the velocity profile, the velocity was determined from four points across the diameter (as per USEPA Method 1 (USEPA 2000b)).

To facilitate sampling for this and other projects, a gantry was constructed to allow the wind tunnel to be moved across the ponds without disturbing the pond surface layer. This gantry consisted of two pontoons constructed from 12” stormwater pipe and a four-metre wide aluminium frame. A full description of the gantry and its design can be found in Casey *et al.* (2002). The gantry is shown in Figure 22.

Figure 22: Wind tunnel and gantry in Use

Odour samples were collected in Melinex™ (Polyethylene Terephthalate) sample bags. The sample bags were placed into a rigid sample container and the air inside the container was evacuated at a controlled rate using 12-volt diaphragm pumps to fill the bags. This is known as the lung method. All components used for sampling

were composed of odour free stainless steel or polytetrafluoroethylene (PTFE) as per the standard.

Prior to sampling the bags were pre-conditioned by filling them with odorous air from the source. After this pre-conditioning, odour samples were collected over a period of six minutes. All odour samples were analysed at the DPI&F olfactometry laboratory on the same day they were collected.

By taking samples across a pond's surface (as shown in Figure 21) the issue of spatial variability could be investigated.

4.5 Assessment of footprint

It has been proposed that, for a sample collected close to a source, whilst the model would allow for the entire source, a limited area of the pond (due to horizontal spread, Figure 9) would unduly influence the modelled predictions (Smith and Kelly 1996; Smith 1995).

To assess the source contribution, each source was broken up into a number of equally sized areas. This breakdown was dependant upon the number of sample points taken across each ponds surface and the shape of the pond. For source, the area within the main source was modelled separately to assess the contribution of the smaller part of the source to the downwind non-dimensional concentration (Ψ) for the entire source. The weighted emission rate was calculated using Equation 16.

$$E_W = \sum_{1}^{n} \left(\frac{\psi_n}{\psi_{Total}} \times E_n \right) \qquad \text{Equation 16}$$

Where:

- n is the number of grid points for the pond;
- E_W is the weighted odour emission rate ($OUm^{-2}s^{-1}$);
- Ψ_n is the non-dimensional concentration for point n on the pond;
- Ψ_{Total} is the sum of all non dimensional concentrations; and

- E_n is the emission rate measured using the wind tunnel for point *n*.

4.6 Assessment of fetch

The fetch theory proposes that the transfer of odour from the liquid surface into the air-stream passing over the surface is reduced as the concentration of odour in the air-stream increases (Nicholas *et al.,* 2003). In an attempt to assess fetch, the distance upwind of each sample site was derived from the data obtained by a GPS survey of the site. The fetch was defined as the point on the pond closest to the sampling point and then upwind to the furthermost point of the opposite pond edge.

To enable the efficacy of the model to be assessed with respect to fetch, the emission rates were expressed as a back calculated odour emission rate minus averaged pond odour emission rate. Thus, if fetch were an issue (based on the proposal of Jiang (2002)), it would be expected that as fetch increases, the back calculated emission rate would decrease thus the majority of points would fall beneath the line.

4.7 Statistical analysis

Linear regressions were performed using Genstat for Windows, Sixth edition.

Chapter 5 Results

5.1 Odour emissions

All odour emissions have been scaled to a standard tunnel wind speed of 1 m/s at half tunnel height using Equation 6.

5.1.1 Piggery A

The results for the three on pond sampling sessions and back calculation modelling for piggery A are shown in Table 6 to Table 8.

Table 6: Results from on pond sampling at Piggery A ($OUm^{-2}s^{-1}$)

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	5.3	10.1	2.8	10.9	10.5	9.5
2	6.5	9.7	39.2	17.4	26.6	7.1
3	6.6	7.0	31.5	8.6	26.6	6.3
4	6.1	8.0	25.9	14.6	39.8	6.0
5	21.3	12.0	8.5	17.4	33.5	6.7
6	11.2	4.4	7.5	17.4	12.5	9.5
7	3.8	11.3	5.7	29.3	13.2	7.8
8	13.0	8.5	4.9	61.3	13.2	5.9
9	13.8	5.3	10.9	16.6	18.8	4.1
Average	9.7	8.5	15.2	21.5	21.6	7.0

Table 7: Concentrations of downwind samples at Piggery A (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	N/A	16	N/A	25	24	22
2	N/A	11	N/A	26	24	25
3	N/A	11	N/A	N/A	N/A	N/A

Table 8: Back calculated odour emission rates for Piggery A ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle[4] (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class[5]		Emission Rate @ 1 m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S[6] 00/01 S1	N/A	143	N/A	1.8	N/A	D[7]	N/A	2.3
S 00/01 S2	N/A	142	N/A	1.9	N/A	C	N/A	2.4
S 00/01 S3	N/A	124	N/A	2.6	N/A	C	N/A	3.4
W 01 S1	N/A	128	N/A	3.4	N/A	D	N/A	10.0
W 01 S2	N/A	37	N/A	4.6	N/A	D	N/A	13.3
S 01/02 S1	135	113	3.3	0.8	C	A	9.4	3.1
S 01/02 S2	126	118	3.9	1.2	C	A	11.3	5.0

5.1.2 Piggery B

The results for the three on pond sampling sessions and back calculation modelling for piggery B are shown in Table 9, Table 10 and Table 11.

+Table 9: Results from on pond sampling at Piggery B ($OUm^{-2}s^{-1}$)

[4] This is not the original value measured by the weather station as it has been adjusted to take into account pond alignment

[5] All stability classes were calculated using Simga-A method.

[6] S = Summer year = 2000-2001 sample number =Sample 1

[7] Neutral stability mid morning during summer may be questionable

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	11.1	11.1	22.2	12.5	32.7	10.3
2	6.1	10.6	12.5	26.6	12.7	7.7
3	7.2	5.2	3.6	9.3	24.3	10.8
4	7	4.5	35.4	8.3	62.2	17.4
5	14.8	6.4	18.9	14.0	50.2	10.9
6	4.2	10.5	3.6	10.7	30.3	10.9
7	N/A	5	7.5	3.1	16.8	17.3
8	2.8	14.9	47.2	8.4	18.0	23.5
9	4.1	9.6	23.6	5.2	76.5	17.3
Average	7.2	8.6	19.4	10.9	36.0	14.0

Table 10: Concentrations of downwind samples at Piggery B (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	N/A	21	12	26	36	31
2	N/A	19	64	17	22	51

Table 11: Back calculated odour emission rates for Piggery B ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class		Emission Rate @ 1 m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S 00/01 S1	N/A	33	N/A	1.7	N/A	A	N/A	5.5
S 00/01 S2	N/A	19	N/A	1.9	N/A	A	N/A	6.2
W 01 S1	287	292	2.7	2.3	C	D	3.6	4.7
W 01 S2	289	304	3.2	2.8	C	D	22.5	4.2
S 01/02 S1	No Data Available due to Meteorological Station Failure							
S 01/02 S2								

5.1.3 Piggery C

The results for the three on pond sampling sessions and back calculation modelling for piggery C are shown in Table 12, Table 13 and Table 14.

Table 12: Results from on pond sampling at Piggery C ($OUm^{-2}s^{-1}$)

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	8.7	14	4.7	5.9	DNS [8]	DNS
2	20.1	18.7	9.8	13.8	DNS	DNS
3	16.7	6.7	8.9	17.2	DNS	DNS
4	2.9	2.5	14.7	10.4	DNS	DNS
5	22.5	8.2	42.7	17.1	DNS	DNS
6	66.7	N/A[9]	35.9	78.7	DNS	DNS
7	30.3	7.8	14	53.3	DNS	DNS
8	28	12.4	27.7	83.7	DNS	DNS
Average	24.5	10.0	19.8	35.0	N/A	N/A

Table 13: Concentrations of downwind samples at Piggery C (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	57	35	45	79	DNS	DNS
2	161	33	30	80	DNS	DNS

[8] No data due to inability to access the site for sampling

[9] Sample analysis was not complete due to split sample bag

Table 14: Back calculated odour emission rates for Piggery C ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class		Emission Rate @ 1 m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S 00/01 S1	64	9	2.4	1.7	A	A	27.8	21.1
S 00/01 S2	57	57	2.8	2.5	B	A	167.6	16.5
W 01 S1	19	43	1.5	3.8	C	D	8.4	29.1
W 01 S2	22	35	2.1	4.6	A	D	11.4	33.3
S 01/02 S1	Did not sample as unable to gain site access							
S 01/02 S2								

5.1.4 Piggery D

The results for the three on pond sampling sessions and back calculation modelling for piggery D are shown in Table 15, Table 16 and Table 17.

Table 15: Results from on pond sampling at Piggery D ($OUm^{-2}s^{-1}$)

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	2.2	1.4	DNS[10]		1.9	39
2	6.9	21.7			19.9	83.9
3	5.6	8.7			28.2	73.7
4	8.2	6.6			28.2	82.7
5	11.4	15.3			29.1	69.5
6	1.7	3.4			25.5	80.3
7	1.7	12.9			22.4	73.7
8	20.7	13.5			27.1	61.9
9	5.1	9.4			19	70.5
Average	7.1	10.3			22.4	70.6

Table 16: Concentrations of downwind samples at Piggery D (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	16	84			78	271
2	22	22	DNS		23	228

[10] DNS – Did not sample due to pond being unavailable due to being desludged

Table 17: Back calculated odour emission rates for Piggery D ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class		Emission Rate @ 1m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S 00/01 S1	68	106	1.5	4.3	B	D	9.1	24.4
S 00/01 S2	87	99	1.8	5.1	A	D	5.4	7.3
W 01 S1 W 01 S2	Did not sample due to the pond being desludged							
S 01/02 S1	111	84	2.3	5.3	D	D	13.9	92.8
S 01/02 S2	141	85	2.5	5.6	D	D	7.7	82.5

5.1.5 Piggery E

The results for the three on pond sampling sessions and back calculation modelling for piggery E are shown in Table 18, Table 19 and Table 20.

Table 18: Results from on pond sampling at Piggery E ($OUm^{-2}s^{-1}$)

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	6.7	5	24.7	3.6	34.0	30.5
2	7.1	17.3	73.1	33.5	41.9	27.9
3	8.7	10.3	7.9	20.9	39.1	33.3
4	10.6	32.3	13.5	30.7	18.2	22.8
5	13.7	9.1	54.8	7.9	24.3	40.7
6	5.8	7.6	34.4	5.8	27.6	39.6
7	10	9.4	27.4	3.4	15.8	61.1
8	21.1	13.5	15.3	7.5	25.8	38.4
9	7.4	10.4	12.1	16.6	12.8	32.3
Average	10.1	12.8	29.2	14.4	26.6	32.3

Table 19: Concentrations of downwind samples at Piggery E (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	28	22	22	N/A [11]	63	101
2	N/A [12]	24	48	42	84	256

[11] Not available: sample not analysed due to problem with olfactometer

[12] Not available: sample not analysed due to problem with olfactometer

Table 20: Back calculated odour emission rates for Piggery E ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class		Emission Rate @ 1 m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S 00/01 S1	190	340	1.9	4.1	A	C	6.8	8.6
S 00/01 S2	N/A	316	N/A	3.2	N/A	C	N/A	7.0
W 01 S1	90	N/A	3.4	N/A	D	N/A	4.6	N/A
W 01 S2	90	225	4.7	3.2	D	A	14.0	16.6
S 01/02 S1	83	98	3.7	2.9	C	D	20.6	18.0
S 01/02 S2	96	107	3.9	2.9	D	D	19.5	50.1

5.1.6 Piggery F

The results for the three on pond sampling sessions and back calculation modelling for piggery F are shown in Table 21, Table 22 and Table 23.

Table 21: Results from on pond sampling at Piggery F ($OUm^{-2}s^{-1}$)

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	DNS	DNS	2	6.8	39.6	10.8
2	DNS	DNS	3.3	7.1	11.6	30.5
3	DNS	DNS	NA	9.3	38	61.1
4	DNS	DNS	NA	6.8	39.6	38.6
5	DNS	DNS	NA	51	25.7	23.3
6	DNS	DNS	NA	66.2	19.8	25
Average	N/A	N/A	2.7	24.5	29.1	31.6

Table 22: Concentrations of downwind samples at Piggery F (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	N/A	N/A	11	22	110	45
2	N/A	N/A	10	78	N/A	73
3	N/A	N/A	14	N/A	N/A	N/A

Table 23: Back calculated odour emission rates for Piggery F ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class		Emission Rate @ 1 m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S 00/01 S1 S 00/01 S2	No samples taken during first sampling period							
W 01 S1	329	280	1.4	2.4	C	C	2.5	14.1
W 01 S2	314	304	2	1.8	C	B	2.6	23.5
W 01 S3	343	N/A	1.4	N/A	A	N/A	3.8	N/A
S 01/02 S1	142	208	1.1	2.4	A	A	21.4	24.3
S 01/02 S2	N/A	209	N/A	1.6	N/A	A	N/A	26.6

5.1.7 Piggery B second sampling sessions

The results for the three on pond sampling sessions and back calculation modelling for second sample sessions at piggery B are shown in Table 24, Table 25 and Table 26.

Table 24: Results from on pond sampling at Piggery B – Second session ($OUm^{-2}s^{-1}$)

Sampling Location	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	9.5	22.7	10.2	17.8	49.5	27.2
2	12.8	7.1	6.7	20.8	35.1	17.0
3	15.7	10.7	14.5	9.2	34.7	34.6
4	18.7	21.6	9.5	17.7	13.0	10.1
5	14.4	8	13.5	19.0	42.7	21.0
6	17.9	20.2	15.5	27.7	29.2	22.8
7	14.2	10.8	7.8	28.4	5.5	36.2
8	15.7	13.3	13.5	19.4	11.5	23.5
9	13.5	20.2	16.7	10.3	9.2	9.1
Average	14.7	15.0	12.0	18.9	25.6	22.4

Table 25: Concentrations of downwind samples at Piggery B – Second session (OU)

Downwind sample number	Summer 2000-2001		Winter 2001		Summer 2001-2002	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
1	N/A[13]	63	24	21	DNS[14]	
2	N/A	76	25	31		

[13] Samples were not taken on this day

[14] DNS- Did not sample due to weather station failure

Table 26: Back calculated odour emission rates for Piggery B – Second session ($OUm^{-2}s^{-1}$)

Downwind Sample Number	Wind angle (from) (Degrees)		Measured Wind Speed (m/s)		Stability Class		Emission Rate @ 1 m/s	
	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2	Day 1	Day 2
S 00/01 S1	N/A	33	N/A	3.4	N/A	B	N/A	30.8
S 00/01 S2	N/A	19	N/A	3.5	N/A	C	N/A	31
W 01 S1	272	322	2.5	3	A	D	8.9	6.4
W 01 S2	279	309	2.6	5.2	A	B	9.5	18.8
S 01/02 S1	Samples were not taken due to meteorological station failure							
S 01/02 S2								

5.2 Confidence Intervals

The confidence intervals show the variability in the analysis that would be associated with using dynamic olfactometry. The 95% confidence intervals for the three seasons are shown in Table 27, Table 28 and Table 29.

Table 27: 95% Confidence interval for summer 2000-2001 olfactometry results

Piggery	Day	Lower ($OUm^{-2}s^{-1}$)	Average ($OUm^{-2}s^{-1}$)	Upper ($OUm^{-2}s^{-1}$)
A	1	7.5	9.7	12.7
	2	6.5	8.5	11.0
B	1	5.4	7.2	9.5
	2	6.6	8.6	11.2
C	1	18.5	24.5	32.4
	2	7.4	10.0	13.6
D	1	5.4	7.1	9.2
	2	7.9	10.3	13.4
E	1	7.8	10.1	13.2
	2	9.8	12.8	16.6
F	Not sampled during this season			
B2	1	11.3	14.7	19.2
	2	11.5	15.0	19.5

Table 28: 95% Confidence interval for winter 2001 olfactometry results

Piggery	Day	Lower ($OUm^{-2}s^{-1}$)	Average ($OUm^{-2}s^{-1}$)	Upper ($OUm^{-2}s^{-1}$)
A	1	11.7	15.2	19.8
	2	16.5	21.5	28.0
B	1	14.9	19.4	25.3
	2	8.4	10.9	14.2
C	1	15.0	19.8	26.2
	2	26.5	35.0	46.3
F	1			
	2	17.8	24.5	33.9
E	1	22.5	29.2	38.1
	2	11.4	14.9	19.3
B2	1	9.2	12.0	15.6
	2	14.5	18.9	24.6

Table 29: 95% Confidence interval for summer 2001-2002 olfactometry results

Piggery	Day	Lower ($OUm^{-2}s^{-1}$)	Average ($OUm^{-2}s^{-1}$)	Upper ($OUm^{-2}s^{-1}$)
A	1	16.6	21.6	28.2
	2	5.4	7.1	9.2
B	1	27.6	36.0	46.8
	2	10.8	14.0	18.2
D	1	17.2	22.4	29.1
	2	54.2	70.6	91.9
F	1	21.0	29.1	40.1
	2	22.8	31.6	43.6
E	1	20.4	26.6	34.7
	2	27.9	36.3	47.3
B2	1	19.7	25.6	33.3
	2	17.2	22.4	29.2

5.3 Summary of odour results

5.3.1 Summer 2000-2001

The on pond and back calculated emission rate data for summer 2000-2001 is shown in Table 30 and Figure 23.

Table 30: Average daily odour emission rates for summer 2000-2001 ($OUm^{-2}s^{-1}$)

Piggery	Day 1		Day 2	
	Average on Pond	Average Downwind	Average on pond	Average downwind
A	9.7	N/A	8.5	2.7
B	7.2	N/A	8.6	6.4
C	24.5	27.8	10.1	18.8
D	7.1	7.3	10.3	15.9
E	10.1	6.8	12.8	7.8
F			DNS[15]	
B2	14.7	N/A	15.0	30.9

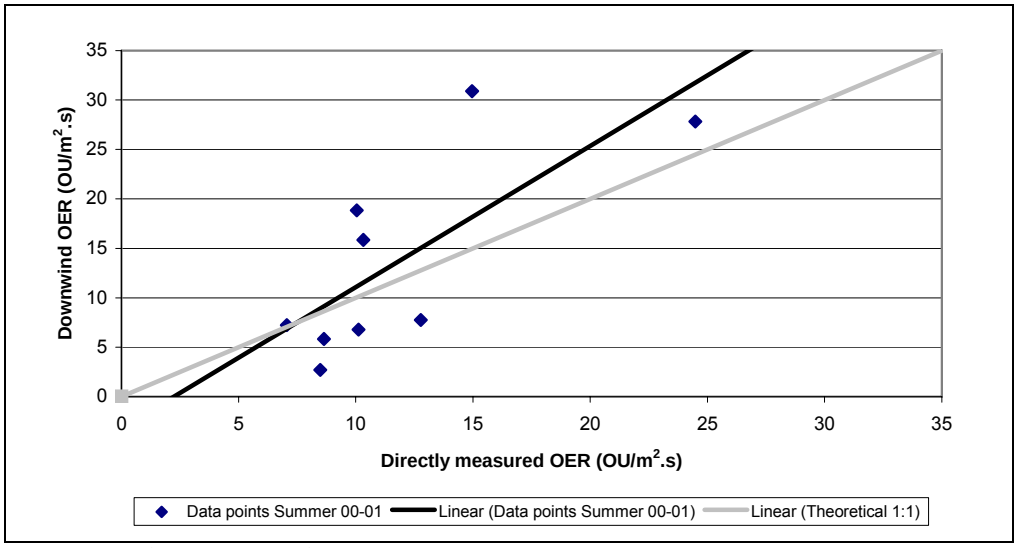

Figure 23: Comparison of on pond and downwind emission rates summer 00/01

5.3.2 Winter 2001

The on pond and back calculated emission rate data for winter 2001 are shown in Table 31 and Figure 24.

[15] DNS – Did not sample Piggery F during this period

Table 31: Average daily odour emission rates for winter 2001 ($OUm^{-2}s^{-1}$)

Piggery	Day 1		Day 2	
	Average On Pond	Average Downwind	Average on pond	Average downwind
A	15.2	N/A	21.5	11.7
B	19.4	13.1	10.9	4.5
C	19.8	9.9	35.0	31.2
D			DNS[16]	
E	29.2	9.3	14.4	16.6
F	N/A	2.8	24.5	18.8
B2	12.0	9.2	18.9	12.6

Figure 24 shows the winter odour emission rate data.

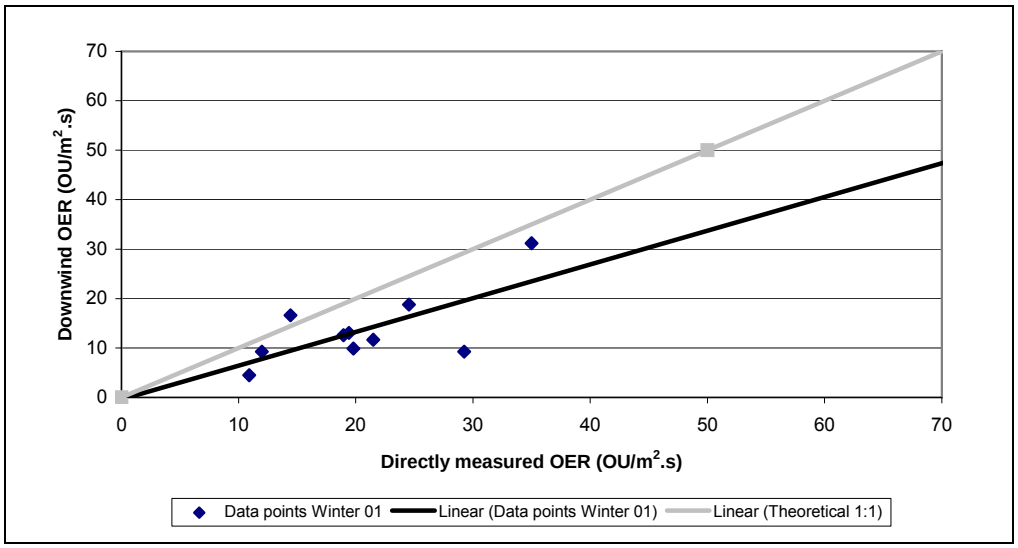

Figure 24: Comparison of on pond and downwind emission rates Winter 2001

[16] DNS – Did not sample PiggerD during this period

5.3.3 Summer 2001-2002

The on pond and back calculated emission rate data for summer 2001-2002 are shown in Table 32 and Figure 25.

Table 32: Average Daily Odour Emission Rates for summer 2001-2002 ($OUm^{-2}s^{-1}$)

Piggery	Day 1		Day 2	
	Average On Pond	Average Downwind	Average on pond	Average downwind
A	21.6	10.4	7.0	4.1
B	36.0	N/A	14.0	N/A
C	DNS[17]			
D	22.4	10.8	70.6	87.7
E	26.6	20.1	32.3	34.1
F	29.1	21.4	31.6	25.5
B2	25.6	N/A	22.4	N/A

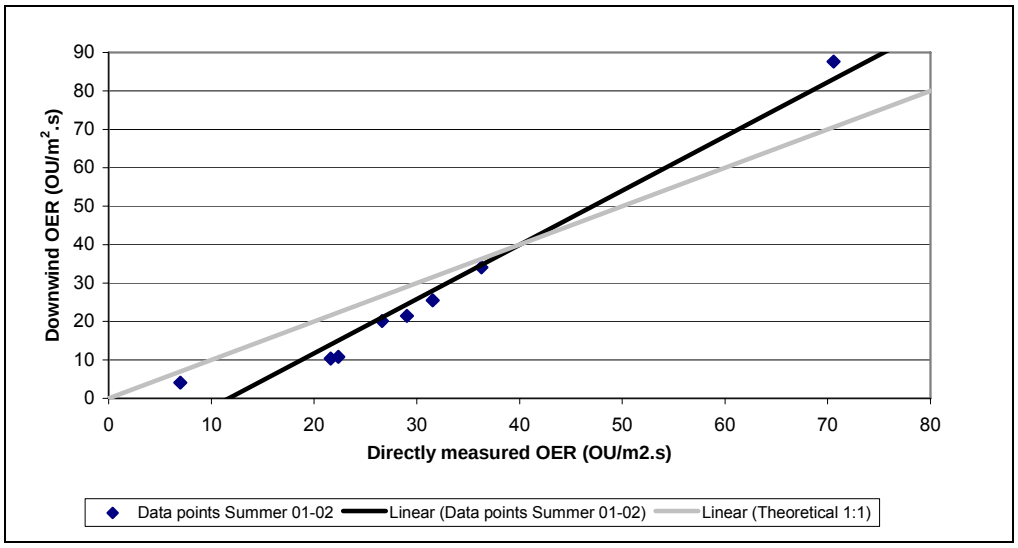

Figure 25: Comparison of on pond and downwind emission rates summer 2001-2002

[17] DNS – Did not sample Piggery C during this period

5.3.4 Overall effectiveness

All pairs of data for downwind modelled odour emission rate and on pond odour emission rate across the three sampling seasons are shown in Figure 26 and the results of linear regressions on the data sets are shown in Table 33.

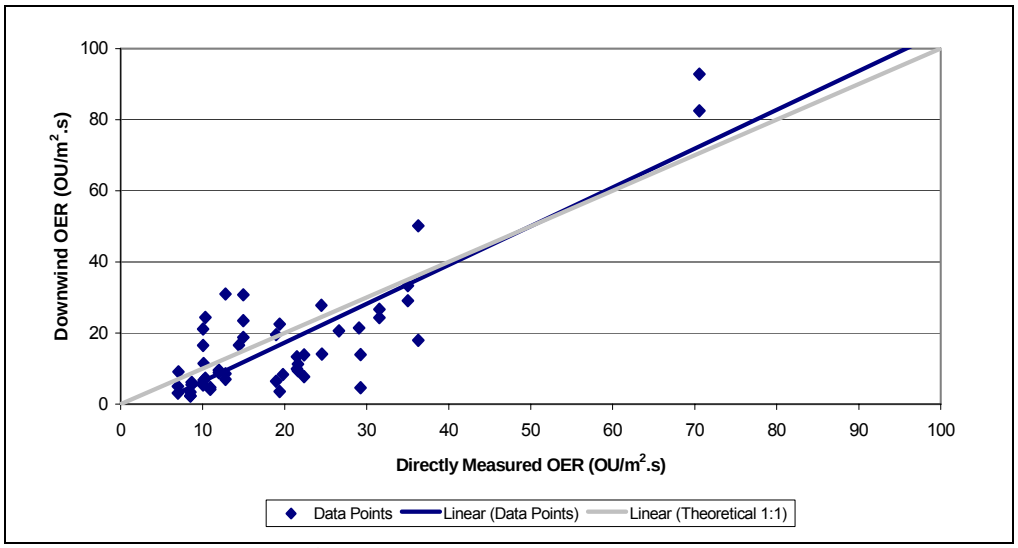

Figure 26: All data points for all seasons

Table 33: Summary of seasonal regression results

Seasons	R^2
Summer 2000-2001	42.5
Winter 2001	45.8
Summer 2001-2002	90.0
All	72.5

5.4 Source footprint

The results of the footprint assessment for the three sampling periods are shown below. Examples of the data required to calculate the influence of footprint is shown in Table 34 and Table 35 below. Both show the modelling of all points and the entire pond for a day at Piggery A and Piggery F respectively. A summary of the footprint

modelling for summer 2000/2001, winter 2001 and summer 2001/2002 are shown in Table 36, Table 37 and Table 38 respectively. The results of the linear regression analysis undertaken on the data sets for the averaged and the weighted emission rates are shown in Table 39.

Table 34: Example of modelling for Piggery A Downwind 1

Source Description	Adjusted Dimensions (m)	Co-ordinates (m from centre)	Ψ	% Total	Direct Measured OER ($OUm^{-2}s^{-1}$)	Weighted contribution to downwind source
Entire	54,47.5	19,31.3	10.33867			
1	18,15.8	3.8,50	0.00003	0.0	10.1	0.00
2	18,15.8	3.8,31.3	0.01835	0.1	9.7	0.01
3	18,15.8	3.8,13.5	3.12689	25.5	7.0	1.79
4	18,15.8	19.5,50	0.08140	0.7	8.0	0.05
5	18,15.8	19.5,31.3	1.82097	14.9	12.0	1.79
6	18,15.8	19.5,13.5	4.06163	33.1	4.4	1.47
7	18,15.8	36,50	0.92511	7.5	11.3	0.85
8	18,15.8	36,31.3	1.99046	16.2	8.5	1.37
9	18,15.8	36,13.5	0.23135	1.9	5.3	0.10
		Total	12.25619	100		7.4

Table 35: Example of modelling for Piggery F - Downwind 1

Source Description	Adjusted Dimensions (m)	Co-ordinates (m from centre)	Ψ	% Total	Direct Measured OER ($OUm^{-2}s^{-1}$)	Weighted contribution to downwind source
Entire	27.5,40	-21,8.8	4.61493			
1	13.5,13.5	-35,16.3	0.28195	5.6	39.6	2.2
2	9.5,9.5	-35,1.8	0.19952	3.9	11.6	0.5
3	13.5,13.5	-21,16.3	0.46054	21.8	38.0	8.3
4	13.5,13.5	-21.5,1.8	1.10471	9.1	39.6	3.6
5	13.5,13.5	-7.8,16.3	0.21481	4.2	25.7	1.1
6	13.5,13.5	-7.8,1.8	2.81007	55.4	19.8	11.0
		Total	5.07160	100		2.2

Table 36: Footprint analysis results – summer 2000/2001

Piggery	Date	Downwind Number	Footprint based OER ($OUm^{-2}s^{-1}$)	Average of all points ($OUm^{-2}s^{-1}$)
A	22/02/01	1	10.1	9.7
A	22/02/01	2	10.1	9.7
A	22/02/01	3	7.5	8.5
B	01/03/01	1	9.7	8.6
B	01/03/01	2	9.0	8.6
C	06/03/01	1	23.5	24.5
C	08/03/01	1	12.6	10.1
C	08/03/01	2	7.3	10.1
D	13/03/01	1	12.9	7.1
D	13/03/01	2	9.2	7.1
D	15/03/01	1	8.2	10.3
D	15/03/01	2	10.4	10.3
E	20/3/01	1	13.5	10.1
E	22/03/01	1	13.5	12.8
E	22/03/01	2	12.3	12.8
B	12/04/01	1	16.6	15.0
B	12/04/01	2	14.8	15.0

Table 37: Footprint investigation results – winter 2001

Piggery	Date	Downwind Number	Footprint based OER ($OUm^{-2}s^{-1}$)	Average of all points ($OUm^{-2}s^{-1}$)
A	26/07/01	1	16.5	21.5
A	26/07/01	2	16.5	21.5
B	17/07/01	1	17.1	19.4
B	17/07/01	2	17.1	19.4
B	19/07/01	1	8.3	10.9
B	19/07/01	2	11.6	10.9
C	31/07/03	1	14.1	19.8
C	31/07/03	2	14.1	19.8
C	02/08/01	1	34.5	35.0
C	02/08/01	2	5.9	35.0
E	14/08/01	1	12.8	29.2
E	14/08/01	2	12.8	29.2
E	16/08/01	1	4.6	14.4
E	16/08/01	2	16.6	14.4
F	09/08/01	1	36.5	24.5
F	09/08/01	2	21.5	24.5
B	21/08/01	1	10.4	12.0
B	21/08/01	2	10.4	12.0
B	23/08/01	1	15.0	18.9
B	23/08/01	2	16.2	18.9

Table 38: Footprint investigation results – summer 2001/2002

Piggery	Date	Downwind Number	Footprint based OER ($OUm^{-2}s^{-1}$)	Average of all points ($OUm^{-2}s^{-1}$)
A	26/02/02	1	33.2	21.6
A	26/02/02	2	33.2	21.6
A	28/02/02	1	6.6	7.0
A	28/02/02	2	6.6	7.0
D	12/03/02	1	25.0	22.4
D	12/03/02	2	24.0	22.4
D	14/03/02	1	74.9	70.6
D	14/03/02	2	74.9	70.6
E	05/03/02	1	25.5	26.6
E	05/03/02	2	29.6	26.6
E	07/03/02	1	28.0	32.3
E	07/03/02	2	28.0	32.3
F	19/02/02	1	33.0	29.1
F	19/02/02	2	29.1	29.1
F	21/02/02	1	24.2	31.6
F	21/02/02	2	24.2	31.6

Table 39: Regression results from comparing back calculated values to weighted and non weighted emission rates

Season	Averaged OER (R^2)	Weighted OER (R^2)
Summer 2000-2001	42.5	26.2
Winter 2001	45.8	19.3
Summer 2001-2002	90.1	80.0
Combined	72.5	67.1

The data in Table 39 shows that allowing for the concept of source footprint and weighting the emission rates accordingly does not improve the correlation between the on pond derived OER and those predicted using the model.

5.5 Stability classes

As shown in the sensitivity analysis, stability class plays an important role in the model's interpretation of odour dispersion. The stability class information obtained during the sampling process is summarised in Table 40. The frequency of occurrence of a particular stability class calculated using the Sigma-A method is shown in Figure 27. The relationship between the measured OERs and the back-calculated OERs sorted by stability class is shown in Figure 28. The average temperatures during sampling for the three sampling periods are shown in Table 41.

Table 40: Number of stability class events during all sampling periods.

Stability class	Summer 2000-2001	Winter 2001	Summer 2001-2002	Total number of occurrences	%
A	7	4	5	16	31
B	2	2	0	4	8
C	5	4	3	12	24
D	3	9	7	19	37
Total	17	19	15	51	100

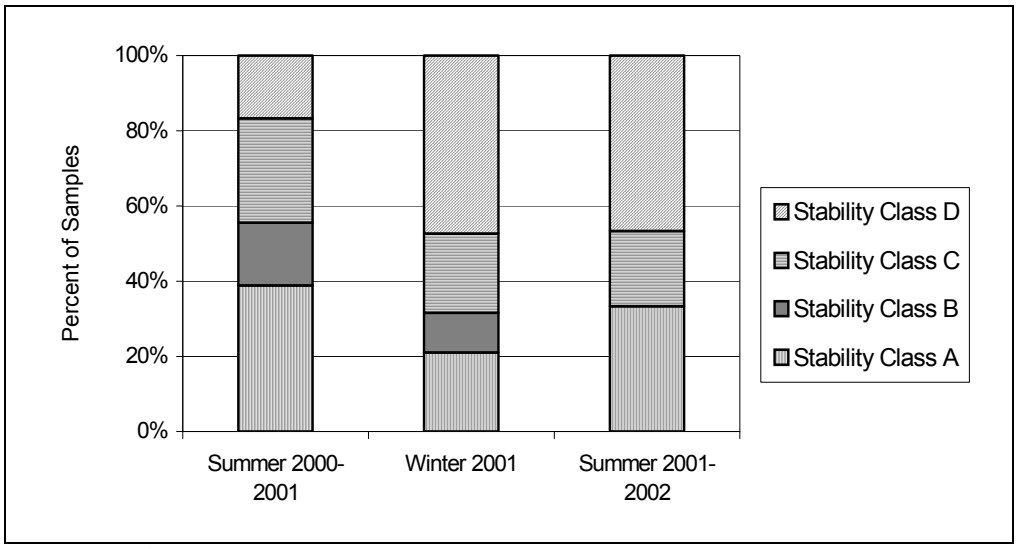

Figure 27: Stability class events during measurement by season

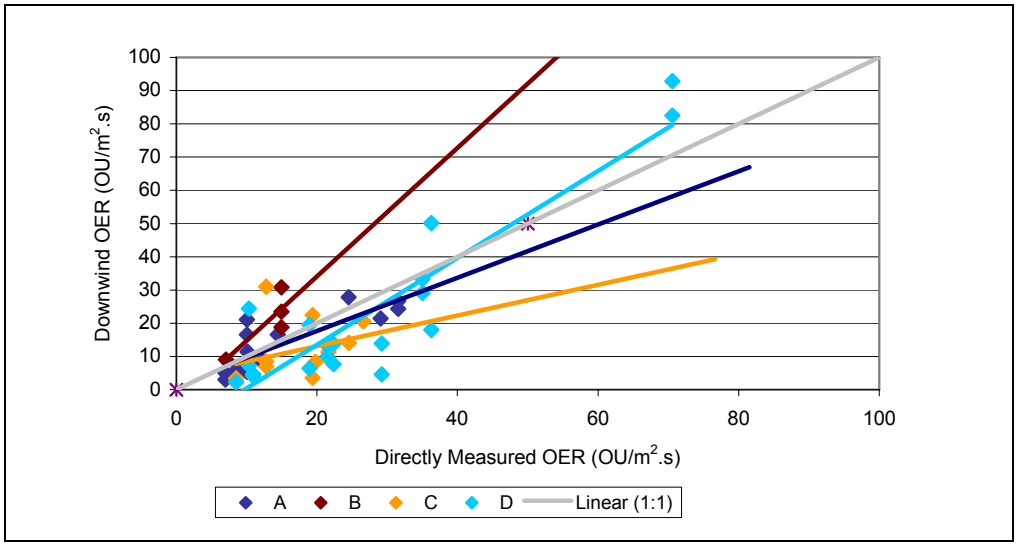

Figure 28: Comparison of STINK prediction and measured OER by stability class

Table 41: Daily average temperatures

Season	Temperature (degrees Celsius)
Summer 2000-2001	26.8
Winter 2001	15.9
Summer 2001-2002	22.5

5.6 Impact of fetch

The results of the fetch analysis are shown in Figure 29. Overall, for all data points for all seasons (Figure 26), 39% of the values had an on pond to downwind ratio of 1:0.7 or better and 61% of values had a ratio of less than 1:0.7.

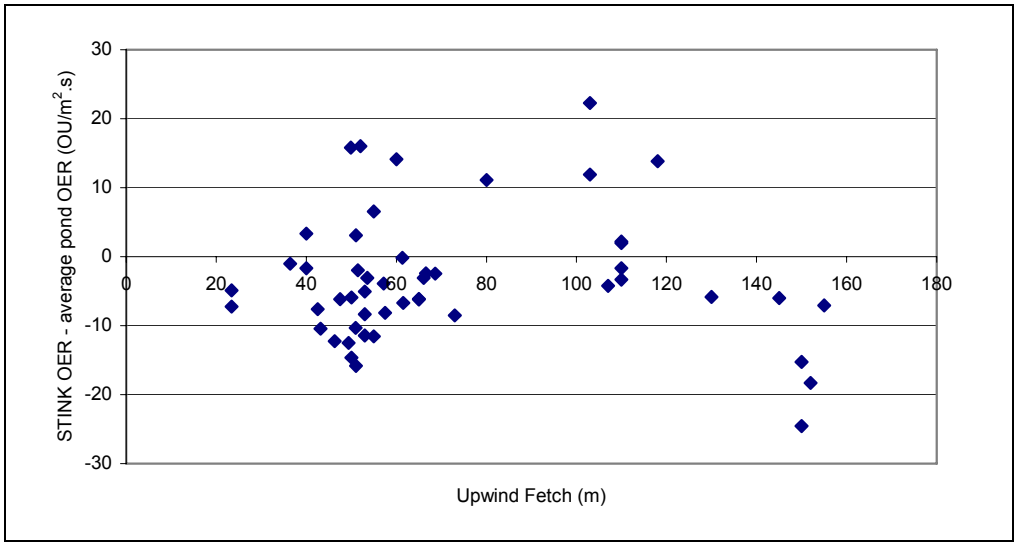

Figure 29: Results of fetch assessment

In Figure 29, there is no evidence of a significant reduction in emission rate, as proposed by Jiang (2002) for ponds up to 120 metres in length.

5.7 Effectiveness on elevated ponds

Of all the ponds studied Piggery B had a pond that was elevated approximately 2 metres above the surrounding landscape. By elevated, it is inferred that the style of pond is that of a “turkey nest” dam and not constructed on a gradient where one or more sides are above the surrounding landscape.

To assess the effect of elevated ponds on the predicted OERs, Piggery B was removed from the data set and the linear regression was calculated on the remaining data using Genstat. The results of this are shown in Table 42.

Table 42: Linear regression results for elevated ponds

Season	With Piggery B	Without Piggery B
Summer 2000-2001	42.5	53.5
Winter 2001	45.8	12.4
Summer 2001-2002	90.1	N/A[18]
All	72.5	80.9

Table 42 shows that overall; removing the elevated pond increased the correlation by approximately 8%. No influence was observed during the summer of 2001-2002 as Piggery B was not modelled due to weather station failure. It is concluded that based on the limited data, using the model on an elevated pond would not significantly alter the models prediction of emission rates.

[18] No difference as due to weather station failure no back calculated values were available for Piggery B

Chapter 6 Discussion

6.1 Introduction

Indirect methods can be used where the collection of downwind odour samples occurs after the ambient air has mixed with odour from the source. This eliminates the chances of errors associated with enclosure methods being introduced. At present, it is unknown which of the available methods actually provides the most accurate estimate of an areal odour emission rate

Downwind methods are used to calculate emission rates from emitting surfaces by combining a concentration measured downwind of the source with local meteorological conditions at the time of sampling. While a number of methods have been used, each method is based upon the premise that a downwind concentration can be related to an upwind source via a mathematical formula (Denmead *et al.,* 1998; Edgar *et al.,* 2002; Koppolu *et al.,* 2002; Sarkar and Hobbs 2003; Smith 1993; Smith and Kelly 1996).

Smith (1993; 1995) developed the STINK model specifically to predict odour dispersion close to a ground level source. The results generated using the STINK model in this report, have shown the model performed moderately well during the summer 2000-2001 and winter 2001 sampling periods. The model performed very well for the summer of 2001-2002.

Overall, the model showed a good correlation (R^2 of 72.5, Table 33) when the three datasets were combined. The odour emission rate calculated using the wind tunnel and the odour emission rate calculated from the STINK model should be similar, that is giving a 1:1 relationship. Figure 26 shows that this was the case, with the relationship being close to 1:1. However, the results from each season shown above do not show this as clearly as the individual season results do not follow the 1:1 line.

Possible causes of this variation would be related to the variation inherent in olfactometry and the limited number of data points used to generate the relationships.

Whether the model over or under predicted emission rates during a given season can be linked back to the relationship between the accuracy and suitability of the methods. That is, which method provides the “real” emissions? The wind tunnel and STINK model have shown a good correlation; from this one may conclude that they both predict the emission rate as well as each other. Previous work, such as that of Sarkar and Hobbs (2003), has shown that back calculation methods are comparable to wind tunnel studies (when using a Lindvall hood). More recent work by Sommer *et al.* (2004) has shown that whilst two downwind methods compared well, when determining gaseous emission rates, neither correlated well with a flux chamber. The lack of data surrounding which of the available methods provides a realistic emission rate hampers further assessments. The overall correlation between downwind and measured values shows promise for future use of the model, and back calculation in general. A number of the variables associated with the use of the modelling are discussed further below.

6.2 Efficacy of model

Both the downwind method and the wind tunnel each provide an estimate of the true emission rate, which currently, cannot be determined directly and accurately. Overall, a good correlation (R^2 of 72.5) was observed for odour emission rates determined using the two techniques. This indicated that the back calculation method and wind tunnel were within a similar order of magnitude and were close to the 1:1 relationship. It is known that wind tunnels and flux chambers do not relate well (Jiang and Kaye 1996) thus the findings of Sommer *et al.* (2004) as discussed previously

reinforced the fact that flux chambers are not suitable for determining emission rates for future use in dispersion models.

A significant difference between this work and previous use of STINK, such as that of Smith and Kelly (1996), is that this project made use of a spatially averaged odour emission rate, (i.e. an average pond odour emission rate derived from a number of discrete samples).

The seasonal results detailed above (Figure 23 to Figure 25), showed a different relationship between the modelled emission rate and that measured directly. The model appeared to over predict the OERs in summer, whilst it under-predicted OERs in the winter period. It is important to note that these results were derived from the limited data points that were available.

Overall, the results for the three seasons (Figure 26) showed that the correlation between wind tunnel and the STINK model were similar to the theoretical 1:1 line. The results of the linear regressions performed on the data were shown in Table 33. Whilst there was enough data to derive a relationship, it can be seen in Figure 26 that the majority of the emission rates are between 10 and 30 $OUm^{-2}s^{-1}$. Obviously, more data points between 30 and 60 $OUm^{-2}s^{-1}$ would benefit this work through an enhanced view of both on pond samples and back calculated odour emission rates.

A number of factors must be considered when explaining the deviation of the relationship from the 1:1 line. The data indicated seasonal variation that could be caused by changes in atmospheric conditions. While the nature of the pond liquor may also change seasonally, both emission rate estimation techniques would presumably be influenced equally.

The sample collection techniques followed for both methods could contribute to the deviation from the ideal relationship. Samples collected from the pond surface were collected as single samples on each sampling day – the variability of emission rate between samples collected over a short time scale remains unknown. This is

important because the odour samples were collected from the pond surface over a period of up to six hours duration. In contrast, the downwind samples were all collected over relatively short periods within this six-hour sampling period.

In conclusion, the correlation between downwind and directly measured odour emission rates shows promise for future use of the model. A number of variables could have influenced the seasonal variation observed and are discussed further below.

6.2.1 Stability class

The stability of the atmosphere refers to its tendency to resist or enhance vertical motion of an air parcel. These were described as follows by Harris *et al.* (1996):

- Stable – A small parcel of air given an upward push tends to return to its original position;
- Neutral – No tendency for a parcel of air to move up or down from its original position; and
- Unstable – A small parcel of air continues to rise after being given a push upwards.

Stability class influences the amount of spread (vertically and horizontally) of the plume and thus the downwind odour concentration. The preferred scheme for describing and categorising atmospheric stability is that proposed by Pasquill in 1961 (USEPA 2000a).

Stability classes for this project were calculated using the on-site meteorological data and varied from very unstable to neutral (as shown in Table 40). Recently there has been discussion over the accuracy and appropriateness of methods used to determine stability classes (eg. Bowen *et al.* (1983), Dewundege (2002), Tripp *et al.* (2004) and Mohan and Siddiqui (1998)). As a result of this, a number of practitioners have made use of a combination of methods to determine a stability class for a particular time period as, often, data collected can vary in quality and quantity.

The results showed that atmospheric stability varied from A to D (very unstable to neutral conditions) during sample collection. The Sigma-A method used in this work is one of the methods listed in the USEPA publication, *Meteorological Monitoring Guidance for Regulatory Modelling Applications.* Due to its widespread use, Turner's method is seen as the best procedure for determining P-G stability (USEPA 2000a) . Of the published literature, very few researchers have used the Sigma-A method. Current work being undertaken by DPI&F at Clifton on the Darling Downs has shown that of 2500 hours, 68% of time the SRDT and Sigma-A method predicted the same stability class (Galvin *et al.* unpublished). Of the other 30% of predicted classes at Clifton, half of these predicted more stable conditions and half predicted unstable conditions.

Experience has shown that under stable conditions, the SRDT and Sigma-A methods had the greatest divergence. Of the samples collected, 37% were collected under stable conditions, 24% under neutral stability and the other 39% were collected under unstable conditions. Thus, in this instance the Sigma-A method is likely to have provided a representative estimation of stability classes and therefore odour emission rate.

The meteorological data quality can influence the selection of a method of calculation of stability class. For this project, the sigma-A method was selected. It is a turbulence-based method, which uses the standard deviation of the wind direction in combination with the average wind speed. It was selected as it was considered that any 2-metre weather station would be able to provide the data required for the calculations. An important, yet sometimes overlooked factor is ambient temperature. It is known that ambient temperatures have a direct link to atmospheric stability (Oke 1975). Changes in temperature create convective currents (Harris *et al.,* 1996), which cause unstable conditions and thus it would be expected that the summer sampling periods would have more occurrences of unstable weather conditions compared to

the winter periods. In addition to the variance caused by stability class, emissions are known to follow Henry's law which, for wastewater treatments plants means that hydrogen sulphide emissions (which can be linked to odour) rise when temperatures increase (Sattler and Devanathan 2004).

Differences of up to 10°C were observed between average ambient temperature of the winter and the summer periods. These differences in temperature could in part explain why the results in Figure 27 did not show a significant difference in stability class events between the seasons. For the winter of 2001 there appears to be an increase of occurrences of stability class D compared to the summer sampling periods. However, an increase in stability class C events was not observed, contrary to expectations. This could be attributable to the time of sampling during the day as unstable conditions generally occur as the ambient temperature increases.

Where possible, downwind sampling was undertaken between 06:00 and 12:00 on the sampling days. For this time of day, the atmospheric stability would be expected to range from very unstable to neutral depending on the meteorological conditions and time of year. Downwind sampling was not undertaken until the onsite wind speeds were at or above 1 m/s as the assumptions underlying Gaussian models do not apply at wind speeds below 0.5 m/s (PAE 2003b). Occurrences of wind speeds greater than 1 m/s generally did not occur until well after sunrise.

Stability classes determined on a sampling day were constant or only varied slightly as expected (i.e. A-B or B-C) between periods over which the downwind samples were collected.

Figure 28 showed that the model performed well under stability classes of A and D. The data indicates that the model performed poorly for stability classes B and C. Large variability was observed for the data set for stability class C. Removal of one of the data points was undertaken after statistical tests indicated it was an outlier.

The revised data set showed a R^2 value for data in stability class C of approximately 42%. This value was closer to those shown for classes A and D in Figure 28.

The cause of the difference in correlation between the two techniques as affected by stability class is unknown as the model allows for variation in stability classes. The difference could be attributed to the number of data points (n≈4-19) or random errors associated with sampling and analysis. Overall, the results observed when the relationship was reviewed in terms of stability class were very similar to those observed for the seasonal results. The results indicate that the model performs in a similar manner irrespective of the stability class. The good correlation over the three seasons indicates that the sigma-A method was appropriate in this instance.

6.2.2 Variability in emissions by sampling day

A one-way ANOVA was conducted to determine whether there was a difference between the two sample days at each of the piggeries for each season. No significant difference was found between the two days for each season. The lack of significance could primarily be the result of no replicates. The power of the test was conducted and it was found to be approximately 20%. This indicates how likely a significant result would be found when it really exists. This is closely related to the number of replicates and the difference between the treatment means with values closer to 80% or above being preferred. A significant result may have occurred, but due to the lack of replication, it has not been found conducting this statistical test.

6.2.3 Impact of pond spatial variability

As seen in Chapter 5, spatial variation was observed in emission rates. If spatial variability were not a real phenomenon, it could be attributed to random errors associated with sampling and analysis. An olfactometer that does not meet the standard could quite easily provide erroneous results. Olfactometry can have a certain level of variability associated with it, however, if an olfactometer is well managed, the level of variability is far less.

The repeatability criterion in the Australian Standard (Standards Australia 2001) is, "*the repeatability (r) shall comply with r ≤ 0.477 (10 ^{r} ≤ 3.0)*". This implies that the factor that expresses the difference between two single measurements, performed on the same testing material in on laboratory under repeatability conditions, will not be larger than a factor of 3 in 95% of cases (Standards Australia 2001). The underlying problem with the back calculation of odour emissions from area sources is the errors inherent in the odour analysis techniques (Sarkar and Hobbs 2003; Smith and Kelly 1995). The variation in samples was assessed using a modification of standard protocol (Sneath and Clarkson 2000; Standards Australia 2001).

If the olfactometer met the repeatability criteria of r≤0.477 (10^{r}≤3.0) a 95% confidence interval can be calculated based on the number of samples taken. The expected variation attributable to olfactometry for a hypothetical 100 OU (Figure 1) showed that as the number of samples increases the expected range of samples analysed decreases.

The effect of variation attributable to olfactometry was assessed on a pond-by-pond basis. As shown in the results, the variation observed is outside of that which could be associated with dynamic olfactometry. This indicated that spatial variability with respect to odour emissions is a real issue when sampling from anaerobic piggery lagoons. This also shows that taking a sample from a single point on a lagoon may not be representative of the spatially averaged OER.

The use of a wind tunnel could be one source of the observed variation. A requirement of the project was to use the wind tunnel to correlate the DPI&F data with data from a project funded by Australian Pork Limited (APL) at the University of New South Wales (UNSW). Additionally the wind tunnel is the preferred method for odour sampling. The tunnel was operated in an identical manner as detailed by Bliss *et al.* (1995). Whilst the internal tunnel velocity may have varied slightly during the sampling process (within the limits described in the aforementioned paper) this would

have been accounted for in Equation 6, which accounts for variations in wind speed. Therefore, it is unlikely that the variation seen could be directly attributable to the wind tunnel. The other possible cause of the variation seen in the results is that associated with the use of dynamic olfactometry for the analysis of odour.

6.2.4 Source footprint

When comparing the results of the direct wind tunnel method and the indirect STINK method it is important to remember that both methods provide only an estimate of the true emission rate, which cannot be determined directly and accurately. The comparison between these two methods has been described previously by Galvin *et al.,* (2004). That study showed a moderate to good correlation (R^2 of 72.5) for the comparison between the emission rates predicted using the STINK model and a directly measured, spatially averaged, odour emission rate (as shown in Figure 26). A significant difference between this work and other less successful work using the STINK model (i.e. Smith and Kelly (1996)), is that this project made use of a spatially averaged odour emission rate. In reviewing area source sampling, Galvin *et al.,* (2003) found only one occasion in the literature (Gholson *et al.,* 1989) where the impact of spatial variability on emissions measurement was addressed.

While the STINK model purports to give an average emission rate for the entire source over the sampling period, this is not strictly correct. With the downwind sampling point being very close to the edge of the source, not all points within the source will contribute equally to the downwind concentration. In fact, the concept of the source footprint (Smith 1995; Flesch 1996), should see particular areas of a pond dominating the concentration at the downwind sampling location. Hence, by weighting the spatially measured odour emission rates according to their contribution to the downwind concentration, it would be expected that the resulting average emission rate would better match that predicted by back calculation. The R^2 of 67.1 suggests that the scatter was greater than without source footprint incorporated, but

the regression line was still close to the 1:1 line. Factors which may mask the footprint effect include temporal variation in emission rates, uncertainties in the definition of the spatial variability of emissions resulting from the limited footprint of the wind tunnel and the small number of measurements, errors associated with the dispersion modelling, and the olfactometry.

It is highly unlikely that the theory underlying dispersion modelling is a significant causal factor. If the Gaussian plume equation was seriously flawed, it would not be an adopted model of regulatory agencies and the good correlation would not have been observed. Let us look at the horizontal dispersion coefficients as an example. The values in Beychok (1994) for Pasquill, Turner, Slade, Gifford and Bowne are all similar in terms of horizontal dispersion for different stability classes. This indicates that multiple studies have confirmed horizontal spread from emitting sources under different stability classes. Furthermore, this indicates that the horizontal dispersion would have been adequately addressed by the model.

The most likely cause of the less than perfect correlations for both the arithmetic and weighted average emission rates is the temporal variation in point emission rates and the time difference between when the two types of samples were taken. Samples collected from the pond surface were collected as single samples on each sampling day. An example of two days sampling for summer 2000-2001 at one pond can be seen in Table 6.

There did not appear to be any reasons for the variation between the two sampling days in the one week. On average, it took 45 minutes to take a sample at a single measurement point, including movement of the wind tunnel and stabilisation prior to sampling. Therefore, the time taken to take the samples at every point on a large pond was approximately six hours. In contrast, the corresponding downwind samples were collected over a relatively short period (6 minutes) within this six-hour period. The correlation between the short-term downwind concentrations and the directly

measured samples showed that the average odour emission rate between the two days was similar, indicating a relatively uniform emission rate with time. However, the points with the highest emission rate on day one did not necessarily have a high emission rate on day two. The emissions vary; however, the cause of this is presently unknown.

A number of studies, other than this study, have examined odour emission rates from ponds (e.g. Heber et al. (2000), Lim et al. (2003) and Smith et al. (1999)), however, at present, none have investigated temporal, or spatial variation, which makes it difficult to compare the data in this document against any other data.

Uncertainties in the definition of the spatial variability of emissions resulting from the limited footprint of the wind tunnel within each sampling grid could also have played a role in the unimproved correlation. The wind tunnel has a footprint (sampling area) of 0.32 m^2. The major assumption within this work was that each point measured at the centre of the sampling grids was representative of the entire surface area within the grid. Rather than indicating that the emission rates vary temporally, the unimproved correlation could simply mean that the small-scale spatial variation in emission rates is more significant. However, the fact that the averaged data and the back calculated emission rates generally follow the 1:1 line suggests that the variation within each grid was no greater than the variation between the grids.

Comment [EM2]: Should this explanation of 'footprint' be mentioned earlier than here?

A further issue with the variation could be evaluated by examining the issues associated with analysis of ambient odour samples by olfactometry. Three of these are (a) the sample decays in the sample bag prior to analysis, (b) the odorous air changes via dilution or other processes as it travels from the source to the sampling location and (c) the concentrations determined using olfactometry are generally above what an ambient concentration would be. The issue of bag materials has been noted in a number of publications (e.g. Hudson *et al.* (2004), Pollock and Friebel (2002), van Harreveld (2003) and more recently Koziel *et al.* (2004)). For the piggery

samples taken in this project, the sample material (Melinex) did not appear to have issues associated with sample stability (Hudson *et al.,* 2004) or other factors including contamination due to manufacturing (Koziel. *et al.,* 2004). All samples were analysed within 12 hours of being taken (which is under half the time recommended by the olfactometry standard), thus, it is unlikely that sample decay is an issue for the sample bag used in this work.

The change of an odour as it travels from a source to the receptor (e.g. through absorption of odorous materials onto plants) is an unanswered issue. One earlier validation study using a Gaussian model concluded that the lack of correlation between measured emissions and predicted emissions through back calculation are a result of interactions between the odorous air and other features prior to reaching a receptor (Edgar *et al.,* 2002). Given that the samples taken in this project were never more than 2 metres from the edge of the emitting surface, it is unlikely that the results would have been affected by this phenomenon (Galvin *et al.,* 2004). The third issue of the downwind sample concentration was also addressed by Galvin *et al.,* (2004). They concluded that the result of error associated with downwind measurement using olfactometry for samples taken close to an emitting surface were not very significant.

The stink model produces a non-dimensional coefficient (Ψ), which is then incorporated into Equation 15 to determine an odour emission rate. In theory, the coefficient for the entire source should equal the sum of the coefficients for the subsets of the source for a common downwind sampling location. As seen in the results (Table 34 and Table 35) this was not always the case. On first glance, it would appear that the model did not provide a similar prediction when the source is divided into a number of smaller sources. What one must remember is that the breakdown of the source was undertaken via scaling of a printed GPS output. The receptor co-ordinates were taken from the scaled plan by measuring the value with a

ruler. As most rulers do not measure distances less than 0.5 mm it is likely that the variation observed was a result of slight variations in the co-ordinate distances associated with scaling from the maps for modelling. Generally, the variation in calculated non-dimensional coefficients did not cause large variations in estimated emission rate, as velocity and concentration both had to be incorporated to calculate an emission rate.

The effect of source footprint has been evaluated for downwind samples taken at piggeries and it has been found that the closest points of the emitting surface do not dominate the measured downwind concentration. This data indicates that whilst the emissions from a piggery vary spatially, they also vary temporally, as the ponds appear to have an “average” odour emission rate with time. While uncertainties regarding spatial and temporal variability still remain, the impact of footprint on the model performance does not appear to be as significant as previously proposed.

6.2.5 Effect of elevation of ponds on model predictions

Roughness height is used to represent the influence of topographic features such as buildings or vegetation (VicEPA 2000). The turbulence associated with the atmosphere around a pond can be linked to roughness elements such as trees and buildings (Harris *et al.,* 1996). This means that any pond elevated above the surrounding ground level would have different turbulence characteristics relative to ponds that were not elevated. For ponds built above the surrounding landscape the use of a standard surface roughness, value may be inappropriate.

The pond at Piggery B was found to be the only pond that was elevated above the surrounding ground level. It was estimated that the top of the pond wall was 2 metres above the surrounding landscape. The impact that results from this elevated source may have had on model prediction was assessed by performing a linear regression on the data sets with and without the data from Piggery B. The results of this assessment were shown in Table 42 above.

The difference in relationship between the emission rates generated using the two techniques with and without data from Piggery B is not large. Removal of data for the elevated pond improved the correlation between the model and wind tunnel predictions indicating that even though the roughness height may not have perfectly described the area near the source, the model still performed well. The improved correlation, however small, does indicate that the use of a standardised roughness height of 0.15m is acceptable in this instance.

6.2.6 Impact of fetch

Jiang (2002) proposed that odour emissions from ponds measured using a UNSW wind tunnel were actually a factor of two higher than those that would be derived from downwind samples. He proposed that fetch caused this effect. The fetch theory proposes that the transfer of odour from the liquid surface into the air-stream passing over the surface is reduced as the concentration of odour in the air-stream increases (Nicholas *et al.,* 2003). This is in effect a consequence of changes to the concentration gradient between the liquid and the atmosphere above it.

The impact of fetch was assessed by determining the distance of each point to the upwind edge of the pond. The difference between the directly measured odour emission rate and the calculated odour emission rate was plotted against the fetch length. It would be anticipated that a consistent bias would indicate that the fetch length might influence the downwind-calculated result. There was no evidence of a trend or bias in the relationship between odour emission rate and fetch length – the data appears randomly scattered over a wide range of fetch lengths and differences in odour emission rate values.

In addition, the similarity between the emission rate results directly measured using the UNSW wind tunnel and those derived from STINK model predictions (thus close to the 1:1 ratio in Figure 29) indicates that fetch cannot be as significant as proposed by Jiang (2002). If fetch were significant, the relationship between measured odour

emission rate and calculated odour emission rate would be highly dependent on the physical pond dimensions or the wind speed.

6.2.7 Difference between seasons

Differences were observed between the three seasons (Figure 23 to Figure 25). Anecdotal evidence from both literature and personal observations has shown that stable conditions are more likely to occur during winter. Thus, it would be expected that the stability classes would be biased towards stable to neutral conditions (D & C) for the winter sampling period. This was not the case. However, as the model considers stability, it would be expected that the model would give an indicative emission rate irrespective of stability class.

For the summer 2000-2001 period the results showed that the model under predicted for the lower emission rates (i.e. < 7 $OUm^{-2}s^{-1}$) and over predicted above this value. For the winter of 2001 the model was found to under predict when compared to the on pond values which may indicate the effects of fetch, however during the summer of 2001-2002 the model again under predicted at around 40 $OUm^{-2}s^{-1}$ and over predicted above this value. This phenomenon could be attributed to the limited number of data points available (n ≈ 25).

There are a number of reasons other than the number of data points to explain the differences in the observed data from the theoretical 1:1 relationship. These include the representativeness of the sampling grid system used, the effects of temperature, the way the stability class was determined including fluctuations in meteorological conditions at the time of sampling.

6.2.8 Measurement characteristics of anemometer and direction vane

Another possible source of variation is that the meteorological data used during this process was unduly influenced by the wind speed and wind direction measurement devices. However, upon inspection it was discovered that the accuracy of the wind speed sensors was 1.5% and the wind direction sensors was 3%. Upon investigation

it was discovered that the 3% accuracy in the measured wind speed would have caused approximately 2% variation in the predicted odour emission rate.

A concern with the meteorological data was that the wind direction could only read between 0 and 355 degrees as the sensor used had a 5-degree dead band from 0 to 355 degrees. This means that values between 0 and 355 degrees would not be registered by the sensor and would default to zero. It should be noted that aligning a anemometer in the field is difficult and aligning and anemometer precisely north may not be a working reality. Thus, issues associated with small changes in wind direction may be out of the research team's control and are unlikely to have significant impacts on the predicted odour emission rates.

The sensitivity analysis identified wind direction as an important input when using the model. Further investigation showed that no readings were used where the wind was in the above-mentioned range thus this would not have affected the results in this document. The accuracy of the sensor was plus or minus 3 degrees. The sensitivity analysis showed that for the pond studied a change in 4 degrees one way resulted in an increase in approximately 10% for the odour emission rate whereas a change the other way resulted in a decrease in the predicted emission rate by 10%.

The sigma-A method uses the standard deviation of the wind direction in combination with the scalar wind speed. The meteorological data used during this project was a combination of all of the data readings over the half hour preceding and post sampling as per USEPA (2000a). Up to 360 readings per 6 minute period were taken by the weather station and thus the use of this averaged data may have reduced the impact or meandering of the weather data and the error inherent in the device.

Chapter 7 Conclusion

The data in this document shows that the STINK back-calculation technique and the UNSW wind tunnel gave similar estimates of odour emission rates. This is important, as the USEPA flux chamber, which has been shown to derive lower emission rates than other methods, is the approved method of the New South Wales Environmental Protection Agency. This may lead to underestimation of odour impacts. As the Gaussian STINK model has been shown to be capable of predicting an odour emission rate with reasonable accuracy it would be expected that a technique that provides a much lower emission rate would not be suitable for odour emission rate determination.

The results of the sensitivity analysis for samples taken close to a source indicate that the input variables in order of importance for the STINK model are:

1. Wind speed;
2. Atmospheric stability;
3. Pond size and alignment;
4. Surface roughness;
5. Wind angle/direction;
6. Averaging time; and
7. Monin-Obukhov length (under stable conditions as the length approaches zero).

Overall, the results show that for samples taken close to an areal source the emission rates calculated using the model are comparable to those derived using multiple samples from a ponds surfaces. These results indicate that:

- If sufficient large samples are taken, the directly measured and STINK model calculated odour emission rates are comparable;

- The removal of the elevated pond from the dataset improved the correlation between on pond measurements and modelling, indicating that the elevation of the source can influence downwind concentrations;
- The similarity between the emission rate results directly measured using the UNSW wind tunnel and those derived from STINK model predictions indicates that fetch does not appear to be a significant effect;
- The model incorporates spatial variability of odour emissions providing and average odour emission rate;
- If two downwind samples are used, the estimate of odour emission rate calculated using the STINK model is likely to be at least as good as that derived from a significant number of samples (say six) collected directly from the pond surface, or better than an estimate derived from a limited number of samples (say two) collected directly from the pond surface;
- While the output of the STINK model is subject to parameter/variable selection, conservative use of input parameters will provide reasonable estimates of emission rate, and
- The STINK model is a cost effective alternative to direct measurement of odour emission rates, particularly when significant numbers of samples are required to be collected directly from a pond surface.

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