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**Assessment of the influence of media particle size on the biofiltration of odorous exhaust
ventilation air from a piggery facility**

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Abstract

Two pilot scale biofiltration systems were constructed and installed at the University College Dublin research farm, Lyons Estate. Experimental units consisting of two pens in a 12 pen pig house were sealed off from other pens. Air from each pen was extracted and treated separately in two biofiltration systems. Wood chips larger than 20 mm were selected as the medium for biofiltration system 1, whereas chips of between 10 and 16 mm were used in biofiltration system 2. The moisture content of the media was maintained at 69±4% (w.w.b.) using a load cell method. The volumetric loading rates ranged from 769 to 1847 m³ [gas] m⁻³ [medium] h⁻¹ over a 63 day experimental period. Both biofilters reduced odour by between 88% and 95%. Ammonia removal efficiencies ranged from 64 to 92% and 69 to 93% for biofiltration systems 1 and 2 respectively. Sulphur-containing compounds were reduced between 9 to 66% and -147 to 51% across biofiltration systems 1 and 2. The pH of the biofilters' leachate remained between 6 and 8. Pressure drop for biofilter 2 was 16 Pa greater than that of biofilter 1 at the maximum volumetric loading rate of 1847 m³ [gas] m⁻³ [medium] h⁻¹. It is recommended that a wood chip media particle size greater than 20 mm be used for large scale operation of a biofiltration system on intensive pig production facilities to reduce the development of anaerobic zones and to minimize pressure drop on the system fans.

Keywords: pig, odour, ammonia, olfactometry, biofilter, abatement.

1. Introduction

Intensive pig production can cause malodorous gases to be formed, which can cause a nuisance in the vicinity of these facilities. An increase in public awareness has led to the stimulation for the development of better legislation to control their release. The main sources of these gases include building ventilation air, manure storage facilities and the spreading of manure. Often these odorous mixtures are a result of animal manure decomposing anaerobically to form unstable intermediate by products resulting in a complex mixture of over 168 volatile compounds of which 30 are odorous (O'Neill and Phillips, 1992b). These compounds resulting from natural biological reactions include organic acids, aldehydes, alcohols, fixed gases, carbonyls, esters, amines, sulphides, mercaptans, aromatics, and nitrogen heterocycles. An important gas is ammonia, which can lead to environmental acidification and the pollution of ground and surface water because of its high deposition velocity (van der Eerden et al., 1998). According to Van der Peet-Schwering et al. (1999), approximately 50% of the ammonia emissions from pig production is from pig housing and manure storage while the other 50% is from land spreading. Significant advances have been made for reducing emissions from land spreading and slurry storage with the introduction of low trajectory techniques and covers respectively (Pain et al., 1991; Li et al., 1997; Moseley et al., 1998; Pahl et al., 2000), but further research is required for the treatment of exhaust ventilation air. Many air pollution control technologies exist for the abatement of odour and volatile organic carbons (VOC's) emissions but installation and operation costs restrict their implementation.

Biological methods are the most cost effective (Vaith et al., 1996) and include biofilters, bioscrubbers and biotrickling filters. Biofiltration is a proven technology as a method of odour and VOC emissions reduction from industrial and commercial sources (Leson and Winer, 1991; Pearson et al., 1991; Goldstein et al., 1996; Vaith et al., 1996; Nicolai et al., 2000). It is a robust, cost effective (in comparison to other technologies) and an efficient means for the treatment of

low concentration biodegradable odorous compounds that are emitted from farm facilities (O'Neill et al., 1992a; Nicolai et al., 1998; Classen et al., 1999; Martinec et al., 2000). The operating principle of a biofilter is that the contaminated air from the building is passed through a filter medium where microorganisms reside. The contaminants in the air transverse to the liquid phase surrounding the biofilm where the microorganisms degrade them to CO₂, H₂O, inorganic salts and biomass (Deshusses et al., 1997). Adequate odour and ammonia reduction of greater than 70% for livestock facilities can be achieved at residence times of between three and fifteen seconds (Rodhe et al., 1986; Zeisig et al., 1987; Classen et al., 1999; Nicolai et al., 2000; Sheridan et al., 2000), but these removal efficiencies depend on the moisture content and characteristics of the media (Cox et al., 1996; Auria et al., 1998; McNevin et al., 1998; Morales et al., 1998; Kim et al., 2000; Krailis et al., 2000; Tawil et al., 2001).

The objective of this study was to assess the influence of two media particle sizes of woodchip (greater than 20 mm and 10 to 16 mm) on the biofiltration of odorous exhaust air from a piggery facility.

2. Materials and methods

2.1. Animal facilities

A pig finishing house on the University College Dublin research farm, Newcastle, Co. Dublin was chosen as the facility on which to set up the experimental units. The building consisted of 12 similar pens with partially slatted floors. Pens 1 and 2 were selected as the most suitable on which to construct the biofiltration systems. Each of the two pens was individually sealed off from the rest of the building and was fitted with an individual variable speed centrifugal fan and thermostatically controlled electrical radiant heater (Fig. 1). Each pen had a separate air inlet. These modifications allowed both pens to be heated and ventilated independently of the other 10

pens. Seven disease-free pigs were placed in each of the pens. All of the pigs used in the trial were Large White by Landrace crossbreed female and were fed manually twice daily with a feed comprising of 46% barley, 20% maize, 29% soya bean meal, 1% tallow and 1% lime. The remaining 3% of the feed comprised of additional minerals.

2.2. Biofiltration system

Air was drawn from pen 1 and 2 by variable speed centrifugal fans and passed through the individual biofiltration systems, each comprising of a humidifier and biofilter as shown in Fig. 1. Pig building ventilation rates vary throughout the year, increasing to a maximum during the warmer summer months in order to maintain the ideal comfort conditions for the animals. For a pig finishing house in temperate climate zones, the recommended winter minimum ventilation rate is approximately $10 \text{ m}^3 \text{ h}^{-1} \text{ pig place}^{-1}$ whereas the recommended summer maximum rate is approximately $100 \text{ m}^3 \text{ h}^{-1} \text{ pig place}^{-1}$ (CIGR, 1992).

A humidifier was incorporated into the design in order to completely saturate the air with water vapour as it enters the biofilter. Air with a relative humidity of less than 100% can result in rapid loss of biodegradation activity, even at 40 to 60% (w.w.b.) water in the medium bed as the incoming gas steadily removes water from the bed by convection and dries out the biofilm first (Auria et al., 1998; Bohn et al., 1999; Krailis et al., 2000). A humidifier was constructed using a cylindrical plastic container with a height of 0.9 m and a diameter of 0.5 m. The humidifier was operated in counter-current flow to obtain maximum air moisturising effect. A humidity/temperature sensor (Model D12-20, Airflow Ltd, UK) at the exit of the humidifier allowed for assessment of the inlet air to the biofilter. A 2 m long pipe with an internal diameter of 0.15 m connected the centrifugal fan to the humidifier. This length of ducting was required to facilitate the accurate determination of airflow rate from the pens (Testo, 1998). A solid cone water nozzle with a spray angle of 80° was placed at the top of the humidifier. Water was

delivered continuously to the atomising nozzle at a rate of approximately 0.6 l min^{-1} by a centrifugal pump. Spent water from the humidifier was piped out at the base. This pipe had a U-shape configuration preventing air from escaping through the water outlet.

2.3. *Biofilter details*

Each biofilter was constructed from a cylindrical metal container with a height of 0.91 m and a diameter of 0.56 m as shown in Sheridan et al. (2000). This allowed for a media depth of 0.5 m, typical for many biofilters (VDI 3477, 1991; Williams and Miller, 1992; Scotford et al., 1996; Wani et al., 1997). The container was fitted with a metal mesh floor 0.125 m from the bottom to support the media. The mesh floor had apertures of approximately 0.015 m. The container was also fitted with a rubber baffle, placed 0.285 m from the top of the containers, and a transparent window to allow viewing of the media. The baffle was approximately 0.075 m wide and was attached around the internal circumference of the containers to prevent short-circuiting of influent gas down the sides of the containers. The temperature within the biofilter media was monitored using two Pt100 platinum thermocouples. Heating tape, covered with fibreglass insulation, was wrapped around the biofilters to provide heat when the average temperature, sensed by the two probes, dropped below a set temperature of 24°C .

The weight of each biofilter was monitored using a load cell method and automated control of moisture content within the biofilters was performed using a solenoid valve, which was controlled using graphical computer programming software called LabView™ (National Instruments, UK). If the weight of the biofilter was known then the moisture content of the biofilters could be controlled to $\pm 4\%$ (Young et al., 1998; Classen et al., 1999).

The contaminated air passed through the media in a down-flow direction as it allowed for easier addition of water directly to the dry zone, which would always occur at the entrance (Arnold et

al., 1997; Krailis et al., 2000). The biofilter weight was recorded every 30 seconds online using the data acquisition software package LabView™.

2.4. *System operation*

The efficiency of the biofilters at reducing odour threshold concentration, odour intensity, ammonia and sulphur emissions from the individual pig pens was assessed over a 63 day experimental period. The biofilter media was inoculated at start up using activated sludge in order to reduce the acclimatisation period (Zeisig et al., 1987; Hamer, 1998; Kim et al., 2000;) and operated for three weeks to allow the biomass to adapt itself to the operating conditions (Degorce-Dumas et al., 1997).

Regular measurements were taken throughout the two systems. These measurements were predominantly generated by sensors located at different areas of the systems and included the internal, external temperature and relative humidity of the individual pens; the relative humidity of the air as it exited the humidifiers and entered the biofilters; the pressure drop across the biofilter medium; the weight and temperature of the biofilters so as to allow the addition of water or heat whenever required via the solenoid valve and the heating tape respectively. The following measurements were carried out manually. These were taken on average every 4 days, so as to allow for acclimatisation of the microbial consortium after each change in airflow rate (Deshusses et al., 1994). These included measurement of the air flow rate; measurement of pH and nitrate concentration of the biofilter leachate, humidifier leachate and the supply water; ammonia concentration entering and exiting the biofiltration systems. Odour threshold concentration, odour intensity and GC analysis were carried out every 7 days.

2.5. *System measurements*

2.5.1. Volumetric loading rate measurements

The volumetric loading rate on a biofilter was considered a key operating parameter and was defined as the volume of gas that passes through per unit volume of filter material per unit time. This was essentially the exhaust ventilation rate of the building divided by the volume of media used to treat this exhaust ventilation. The ventilation rate of the building was determined using a 16 mm vane anemometer (Testo, UK), which was placed in the straight section of ducting between the centrifugal fan and humidifier.

During the 63 day experimental period, the volumetric air loading on the biofilters was increased by a total number of 8 steps. The initial load on the biofilters was $1180 \text{ m}^3 \text{ [gas] m}^{-3} \text{ [medium] h}^{-1}$, at which it remained constant for 21 days. Once odour and ammonia concentration measurements were taken before and after the biofiltration systems for this volumetric loading rate, the volumetric load was increased. The same procedure was repeated over the course of the experiment up to a maximum volumetric loading rate of $1847 \text{ m}^3 \text{ [gas] m}^{-3} \text{ [medium] h}^{-1}$. When the maximum volumetric loading rate was reached, it was then reduced to a minimum of $769 \text{ m}^3 \text{ [gas] m}^{-3} \text{ [medium] h}^{-1}$ and increased again to $923 \text{ m}^3 \text{ [gas] m}^{-3} \text{ [medium] h}^{-1}$. This was performed to examine the biofilter under dynamic conditions to which it would be subjected in a large scale situation and to determine optimum operational conditions. These volumetric loading rates were selected as it operated in the approximate range of 720, 800, 1600 and 2144 $\text{m}^3 \text{ [gas] m}^{-3} \text{ [medium] h}^{-1}$, performed by Janni et al. (2000), Zeisig et al. (1987), Mannebeck et al. (1994) and Siemers et al. (1997) respectively on biofiltration systems that treated emissions from pig and poultry housing.

2.5.2. Collection of the odour samples

In order to transport air samples to the laboratory for odour and GC assessment, a static sampling method was used where air samples were collected in 8.5 litre Nalophan bags using a vacuum sampling device that operates on the 'lung principle', whereby the air is removed from a

rigid container around the bag by a battery powered vacuum pump. This caused the bag to fill through a stainless steel tube whose inlet is placed in the odour stream, with a volume of sample equal to the volume of air evacuated from the rigid container. Air samples were taken before and after the biofiltration systems and maintained under appropriate conditions prior to measurement so as not to alter the odorous gas mixture. Samples were tested within 6 hours of collection as sample deterioration was presumed to be approximately 1% per stored hour (Jiang, 2000).

2.5.3. *Measurement of odour threshold concentration*

An ECOMA TO7 yes/no olfactometer was used throughout the experimental period to determine the odour threshold concentration of the ventilation air before and after biofiltration. The odour threshold concentration was defined as the dilution factor at which 50% of the panel could just detect the odour. Only those panel members who passed screening tests with n-butanol (certified reference gas, CAS 72-36-3) and who adhered to the code of behaviour were selected as panellists for olfactometry measurements (CEN, 1999).

The odour threshold concentration was calculated according to the response of the panel members and was displayed in $\text{Ou}_E \text{ m}^{-3}$, which referred to the physiological response from the panel equivalent to that elicited by 40ppb/v n-butanol evaporated in one cubic metre of neutral gas (CEN, 1999). Odour units were considered a dimensionless unit, but the pseudo-dimensions of $\text{Ou}_E \text{ m}^{-3}$ have been commonly used for odour dispersion modelling, in place of 'grams m^{-3} ' (McGinley et al., 2000).

2.5.4. *Measurement of odour intensity*

Odour intensity was the perceived 'strength' of an odour above its threshold concentration and was expressed as odour intensity levels, which were verbal descriptions of an odour sensation to which numerical values were assigned (Misselbrook et al., 1993; Jiang and Sands, 1998; Jiang et

al., 2000). Stevens proposed that the perceived psychological intensity is a function of the odourant concentration.

$$I = k(C-C_0)^n \quad (1)$$

Where: I: perceived psychological intensity; k: constant dependent on the choice of units of C, I and odourant; C: physical intensity expressed as concentration of odour compound; C_0 : an estimate of the odour detection threshold; n: constant dependent on odourant (Callan et al., 1993).

Eight panel members were required for the determination of odour intensity, which adhered to the VDI 3882 guidelines (1992). When the first four panel members finished the test sequence, the second four began. The Z_{50} was entered into the computer and it calculated the selection range and determined by way of random selection the sequence of steps. The panel members were presented with a sample of odorous gas from which they had one minute to evaluate if the sample was in the range from 'not perceptible (0)<very weak odour (1)<weak odour (2)<distinct odour (3)<strong odour (4)<very strong odour (5)<extremely strong odour (6)'. The samples were presented to the panellists in a random sequence of steps of concentration. When the sequence was completed, the results were presented in tabular form. The odour strength would then be characterised from its mean intensity score.

2.5.5. *Measurement of ammonia concentration*

Ammonia analysis was performed before and after biofiltration systems 1 and 2 using electrochemical cells (7AM CitiCeL, Citytech, UK.) and data was logged online using the software package LabViewTM. The electrochemical cells had a measuring range of 0 to 50 ppm. They were connected to the fieldpoint modules of LabViewTM and when placed in the waste air stream, the ammonia concentration was recorded on a computer. The sensors were pre-calibrated using standard gaseous ammonia concentrations (Citytech, UK). After stabilisation of the sensor,

a sampling time of 30 minutes was used to determine the ammonia concentration in the waste air stream. Sensor resolution was 0.5 ppm.

2.5.6. *Gas chromatography analysis*

The percentage removal efficiency of sulphur-containing compounds across biofiltration systems 1 and 2 was examined using a Varian 3800 gas chromatograph (JVA Analytical, Dublin). Due to the low concentrations of odorous sulphur compounds in the waste air of intensive pig facilities, a pre-concentration step was incorporated. Some 400 ml of waste gas was pre-concentrated on Tenax GR at 20 ml/min for 20 minutes. The Tenax GR was cooled to 4°C to enhance the trapping efficiency of very volatile compounds. When trapping was complete, an automated 6 port valve switched lines to allow carrier gas to be swept through the pre-concentration trap. The trap was heated instantaneously to 290°C. The compounds were desorbed and swept on to a DB624 70-metre capillary column. The column was temperature programmed as follows; 30°C for 42 minutes, 150°C at 7°C min⁻¹, 200°C at 15°C min⁻¹, 250°C at 20°C min⁻¹ and maintained at this temperature for 17 minutes. The total run time was 82 minutes. The carrier gas flow was 1.8 ml min⁻¹ helium and detection was performed with a PFPD specifically for sulphur compound detection.

2.5.7. *pH and Nitrate measurements*

Samples of biofilter leachate, humidifier leachate and supply water were returned to the laboratory for pH and nitrate analysis on average every four days. Three pH buffer solutions; four, seven and nine were used to calibrate a Russell combination pH electrode which was linked to an Orion 520A pH/mV meter. An electrode slope was produced and the pH of each of the samples was determined. A nitrate ion selective probe (Model Orion, 9707 ionplus) was also attached to an Orion 520A pH/mV meter. A standard curve was developed for nitrate concentration determination using nitrate standards at a 10 fold increasing concentration, to

remove any inaccuracies due to sensor drift. When the electrode was placed in the solution a millivolt reading was produced which was then converted into ppm using the software package Microsoft Excel™.

2.5.8. *Statistical analysis*

Statistical analysis was performed according to Wheater et al. (2000) and Mason et al. (1996). Probability (P) values, standard deviations (S.D.), standard errors (S.E.), and R² correlation coefficients were calculated using the software packages Sigmaplot 4.0™ (SPSS Inc., USA) and Microsoft Excel™.

3. Results and discussion

3.1. Odour and ammonia emissions from pen 1 and 2

The geometric mean odour concentrations measured in pens 1 and 2 were 829 and 859 Ou_E m⁻³ respectively. The average odour concentrations from each pen were not significantly different (P = 0.81) from each other; therefore mass odour loading (Ou_E s⁻¹ LU⁻¹) on biofiltration systems 1 and 2 was similar. The geometric mean odour emission rate measured in pens 1 and 2 was 42 and 44 Ou_E s⁻¹ LU⁻¹ respectively for partially slatted floors of 21% of the total floor area. The geometric mean odour emission rate for pens 1 and 2 were not significantly different (P = 0.90) and was comparable to that of other researchers. Martinec et al. (1998) found a wide range of published data describing odour releases from fattening pigs and the odour emissions reported varied between 38 and 495 Ou_E s⁻¹ LU⁻¹ for fully slatted floors. Hartung et al. (1998), Heber et al. (1998) and EPA (2000) reported odour emission rates of 126, 96 and 108 Ou_E s⁻¹ LU⁻¹ respectively from fully slatted floor housing systems. Holste et al. (1998) reported an odour emission rate of 39 Ou_E s⁻¹ LU⁻¹ from a partially slatted floor housing system and Muller et al. (1994) reported an odour emission rate of between 32.8 and 58.8 Ou_E s⁻¹ LU⁻¹ (unspecified floor

type). The values of 42 and 44 $\text{Ou}_E \text{ s}^{-1} \text{ LU}^{-1}$ were similar to the figures of Muller et al. (1994), Holste et al. (1998) and Martinec et al. (1998) but lower than that of Hartung et al. (1998), Heber et al. (1998) and EPA (2000). This can be explained as the odour emission rates from pens 1 and 2 were from a pig fattening house with partially slatted floors while those obtained by the other researchers were from fully slatted floors. It has been suggested by researchers that inside temperature, slatted surface area, ventilation rate, management of the pig facility and feeding regimes significantly affect odour emission rates from pig facilities (Hartung et al., 1998; Heber et al., 1998; Schauburger et al., 1999).

The arithmetic mean ammonia emission rate from pens 1 and 2 was 9.9 ± 2.5 and 8.8 ± 3.6 $\text{g animal}^{-1} \text{ day}^{-1}$ respectively; therefore, there was no significant difference ($P = 0.33$) between the pens. The mass ammonia emission was comparable to that of other researchers. Hendricks et al. (1999) estimated that fattening pigs produce $8.22 \text{ g animal}^{-1} \text{ day}^{-1}$. Demmers et al. (1999) reported mass ammonia emission rates of 4.4, 6.5, 7.4, 7.7 and $9.2 \text{ g animal}^{-1} \text{ day}^{-1}$. Aarnink et al. (1998) determined a mass ammonia emission of $6.84 \text{ g animal}^{-1} \text{ day}^{-1}$. Plots of mass ammonia emission of pens 1 and 2 versus airflow are presented in Fig. 3. As airflow increased, overall mass ammonia emission rate increased (R^2 correlation coefficient 0.41 and 0.74 for pens 1 and 2 respectively). There was slight soiling of the solid floor in pen 1. Krause et al. (1991) reported that highest ammonia concentrations were present at 0.6 metres above floor height. Aarnink et al. (1998) demonstrated that a ventilation rate change of $0.01 \text{ m}^3 \text{ s}^{-1}$ per m^2 floor area can cause a 21.2% change in ammonia emission and that air velocity above the slurry in the pit had substantial effects on ammonia emission. As the ventilation fans were situated one metre above the slatted area, the increase in airflow rate probably caused an increased air velocity within pens 1 and 2 and hence enhanced ammonia volatilisation from the floors and slurry surfaces. Temperature, pH and ammoniacal nitrogen content of the slurry significantly affect ammonia volatilisation. The average temperature in both pen 1 and pen 2 was in the range 18-23°C.

3.2. System performance in terms of odour removal

Biofiltration systems 1 and 2 performed well at treating odour emissions being emitted from the exhaust air of the pig pens. Both systems attained average efficiencies of greater than 92% (Fig. 4). This was comparable with the removal efficiencies of 71.5%, 81%, and 90% attained by Janni et al. (2000), Martinec et al. (2000), and Sheridan et al. (2000) respectively. It was apparent that odour emission rate increased after the biofilters as volumetric loading rate increases (Fig. 4). There was no decreases in odour removal up to a volumetric loading rate of $1693 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$; therefore the odour removal efficiencies of biofiltration systems 1 and 2 are mainly influenced by the odour concentration in the influent gas at the entrance of the biofilters (Hartung et al., 1998). Rihn et al. (1997) reported that increasing nitrate concentration within a trickling bed biofilter lead to the development of a more stable operation and enhanced removal at high inlet concentrations of ethers. Morales et al. (1998) demonstrated that the addition of gaseous ammonia in the inlet stream saturated with toluene increased its elimination capacity noticeably. Gribbins and Loehr (1998) reported that soluble nitrogen concentration in the medium can limit biofilter performance after long periods of operation and even at low inlet VOC loading rates, the biofilter requires a threshold amount of soluble nitrogen to maintain pseudo steady-state operation. Since ammonia and nitrates are present in biofilters operating on pig facilities (Janni et al., 2000; Martinec et al., 2000; Sheridan et al., 2000), it can be suggested that a greater removal efficiency can be obtained for these reasons. Deshusses et al. (1997) suggested that water-soluble compounds were first adsorbed/absorbed on to the packing material and then biodegraded.

There is no significant difference ($P = 0.86$) between biofiltration systems 1 (greater than 20 mm) and 2 (10 to 16 mm) for odour elimination capacity with both systems performing similarly (Fig. 5). Odour elimination capacity was influenced by inlet odour concentration (Fig. 6). This was also demonstrated by Martinec et al. (2000). It is important to note that the odour character of the filtered air was earthy up to a volumetric loading rate of $1693 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$

and resembled a slight piggery and ammoniacal odour only at volumetric loading rates of 1847 m^3 [gas] m^{-3} [medium] h^{-1} and greater.

3.3. *System performance in terms of odour intensity*

Upon investigation of the intensity variation of the inlet and outlet air of biofiltration systems 1 and 2, the slope of the intensity curves of each emission could be distinguished from each other. Misselbrook et al. (1993) also observed this relationship for emissions from pig slurry and broiler housing. The relationship relating odour concentration and intensity from data obtained from pens 1 and 2 combined and the exhaust of biofiltration systems 1 and 2 combined respectively, are illustrated in Figs. 7 and 8. The overall data set for odour intensity relationships is reported in Table 1; it can be seen that the slope coefficient of odour from pens 1 and 2 differ from the slope coefficient after biofiltration systems 1 and 2. As the slope coefficient (n value in Equation 1) increases, the intensity experienced for a given change in concentration changes more considerably and therefore the odour is more likely to be experienced as intense as the concentration increases and less intense as the concentration decreases (VDI Guidelines, 1992). In terms of odour abatement, the odourant with the lowest slope coefficient can be greatly diluted without greatly affecting the odour perception (i.e. exhaust ventilation air from the pig facility) while dilution of an odourant with a high slope coefficient results in rapid dissipation of the odour (i.e. exhaust from biofiltration systems) (Callan et al., 1993). Both biofiltration systems 1 and 2 were effective at increasing the slope coefficient, with biofiltration system 2 being 4% more effective at increasing slope coefficient than biofiltration system 1 (Table 1).

3.4. *System performance in terms of ammonia removal*

Since the biofilter media is maintained at a moisture content of approximately 69%, it acts as a biosorbent. As ammonia has a high Henry's constant, it is easily absorbed on to the packing

material of the biofilter, forming ammonium. The bacteria within the biofilm utilise the ammonium and degrade it to nitrite with the production of H^+ ions, reducing the pH of the system. This increases the capacity of the biofilter to adsorb and absorb more ammonia, therefore increasing the pH. Simultaneously, nitrite is being degraded to nitrate. According to Anthonissen (1976), the nitrification reactions are mainly inhibited by unionised ammonia (FA) and free nitrous acid (FNA).

Martin et al. (1996) suggested increasing CO_2 concentration in the feed gas increased ammonia elimination capacity. As waste air from a pig facility contains between 600 to 1000 ppm CO_2 concentration approximately (Ni et al., 1998; Campbell et al., 1999; Martinec et al., 2000), this waste gas is ideal for the biofiltration of ammonia gas.

Bioscrubbers operated at retention times of approximately 1.5 to 2.7 seconds have achieved ammonia removal efficiencies of approximately 90 to 95% (Schirz et al., 1977; Scholtens et al., 1989). This retention time is sufficient for the transfer of ammonia from the gas phase to the liquid phase where it is biodegraded to nitrite and finally to nitrate. Sublette and Sylvester (1987) indicated that microorganisms could metabolise gas within a short time (several seconds). Deshusses et al. (1994) suggested a minimum retention time of 2 seconds for biodegradation. Hagopian et al. (1998) reported that nitrifiers demonstrate a markedly greater survival rate (approximately 10 times) when attached to a particle as compared to being unattached and Scholtens et al. (1989) suggested that biofilters have a greater capacity for converting ammonia to nitrate when compared to bioscrubbers.

Biofiltration systems 1 and 2 were very efficient at removing ammonia with average removal efficiencies of 79% (64%-92%) and 81% (69%-93%) respectively over the experimental period (Fig. 9). The initial activity of both biofilters was low in relation to the production of nitrate (Figs. 10 and 11). The pH of the leachate was 7.6 and 7.7 for biofilters 1 and 2 respectively, demonstrating the adsorption and absorption of ammonia. As mass ammonia loading increased, nitrate concentration in the leachate increased. There was a slow decrease in the pH of the

leachate showing the production of nitrite, which was simultaneously degraded to nitrate. This production of nitrite increased the absorbance capacity of the media for ammonia. When the volumetric loading rate was reduced to $769 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$ for both biofilters, the mass ammonia loading decreased by 58% and 71% respectively. Percentage efficiency for both biofiltration systems fully recovered (Fig. 9). At a volumetric loading rate of $923 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$, there was a drastic drop in removal efficiency to 69% and 74% respectively for biofiltration systems 1 and 2. This was due to a fault in the water sprinkler system and the media moisture content decreased approximately 16% (w.w.b.). Thus, maintaining a proper moisture content within the biofilter beds is important. The mass ammonia emission exiting biofiltration systems 1 and 2 (Fig. 12) apart from volumetric loading rate $923 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$, increased as volumetric loading rate increased. As volumetric loading rate increased, retention time decreased, therefore causing an increase in the mass emission rate from both biofilters as contact time between the waste air and filter medium may have been both diffusion and reaction rate limiting. Chung et al. (1997), Martinec et al. (2000), and Sheridan et al. (2000) demonstrated that increasing volumetric loading rate affects the removal efficiency of ammonia in biofiltration systems. Janni et al. (2000) and Sun et al. (2000) suggested that medium moisture content significantly affected the removal of ammonia in biofilters and demonstrated that biofilters with a higher moisture content had a greater removal efficiency of ammonia. Colanbeen et al. (1992) demonstrated ammonia removal of between 56 to 98% in a biofiltration system operated at a retention time of 4 seconds and a medium moisture content of 65%. Janni et al. (2000) reported ammonia removal efficiency up to 67% on a biofiltration system operated at a retention time of 5 seconds. Kim et al. (2000) reported that organic packing materials had a higher maximum removal rate than inorganic packing materials for ammonia.

2.6. System performance in terms of reducing sulphur containing compounds

Up to 8 and 14 sulphur containing compounds were detected in the waste gas stream of pens 1 and 2 respectively. Six of the ten most odourous compounds with the lowest odour detection threshold contain sulphur (O'Neill and Phillips et al., 1992). Biofiltration system 1 achieved maximum removal efficiencies up to 65% with a minimum efficiency of 8% (Table 2). Biofiltration system 2 achieved a maximum removal efficiency of 50% and a minimum removal of -147%. Sun et al. (2000) reported that a biofilter operated at a retention time of 5 seconds and a moisture content of 50% (w.w.b.) removed 47 to 94% of hydrogen sulphide in the waste gas stream. She also reported that percentage moisture content significantly affected total sulphur accumulation on the filter medium.

On two measurement occasions, overall sulphur emissions from biofiltration system 2 were increased (Table 2). As the medium packing density was higher and void volume was lower in biofiltration system 2 (see section 3.5.), this allowed for the development of anaerobic zones (excess interstitial water) within the biofilter bed, where organisms can create sulphide and sulphur-containing organics (Devinny et al., 1999). The overall sulphur removing capacity of each bed was also demonstrated in the intensity results where the exhaust air from biofiltration system 2 was more intense than that of biofiltration system 1.

2.7. System performance in terms of pressure drop

Although there was an increase in overall odour and ammonia removal using smaller wood chips (10 to 16 mm), there was also an increase in pressure drop compared to that with wood chips greater than 20 mm. The pressure drop difference is linear to increasing airflow loading on both biofiltration systems (Fig. 13). There was a greater flow resistance in biofilter 2 due to lower void volume (60% compared to 64% as per Hodge and Devinny (1995)) and greater surface area for attachment of biofilm ($146 \text{ m}^2 \text{ m}^{-3}$ - $234 \text{ m}^2 \text{ m}^{-3}$ compared to $86 \text{ m}^2 \text{ m}^{-3}$ - $108 \text{ m}^2 \text{ m}^{-3}$ as per Bibeau et al. (2000)). No sudden change in pressure drop occurred across biofilter

medium 1 or 2 for any of the volumetric loading rates throughout the course of the experiment. This demonstrated that the media had good mechanical strength that led to negligible bed compaction and avoided short-circuiting during operation. Other researchers concluded that wood chip offers the most economically acceptable option with excellent stability properties even after wetting (Phillips et al., 1995). When compared to other packings such as compost, peat and coconut fibre, the pressure drop across wood chip is minimal and will reduce overall power consumption for operation of biofiltration systems (Phillips et al., 1995; Martinec et al., 2000).

2.8. pH of the inlet and leachate water

The pH of the biofilter leachate and the water used to sprinkle the filter bed was determined on each day of ammonia measurement. The pH of the leachate fluctuated between 6 and 8. According to Swanson and Loehr (1997), this range is optimal for biofiltration processes, therefore it was not necessary to add buffer solutions to the pilot scale filter beds.

This fluctuation was due to the production of acid intermediates (i.e. nitric acid from the degradation of ammonia and sulphuric acid from the degradation of hydrogen sulphide and sulphur organics). As microbial species are sensitive to rapid pH fluctuation (Devinny et al., 1999), it may be necessary to add buffering agents such as pelleted calcium carbonate/calcium magnesium carbonate (Demmers et al., 1993) for large scale operation.

4. Conclusions

A pilot scale biofiltration system containing two identical biofilters was designed and built in the University College Dublin research farm, Newcastle, Co. Dublin. Wood chips were used as the media type with chip size of greater than 20 mm and 10 to 16 mm being used in biofiltration systems 1 and 2 respectively. The data acquisition system LabView™ was used to

record system data on-line and olfactometry, electrochemical cells and gas chromatography were used to determine odour threshold concentration, intensity slope coefficient, ammonia concentration and sulphur compound removal efficiency.

Both biofiltration systems 1 and 2 achieved odour reduction efficiencies in the range 88% to 95%, but as volumetric loading rate increased, overall odour emissions from both biofilters increased. Upon examination of the odour intensity concentration relationship, it was found that biofiltration system 2 increased the slope coefficient 4% greater than biofiltration system 1. Both biofiltration systems 1 and 2 achieved comparable ammonia removal efficiencies of 79% and 81%. Thus, biofiltration is an effective technology for the removal of odour and ammonia from the exhaust ventilation air of pig rearing facilities.

Investigation of sulphur compound removal efficiencies demonstrated that it was more likely that biofiltration system 2 would develop anaerobic zones within the medium bed, which would lead to increased emissions of sulphur-containing compounds and thus could cause operational problems in large scale operations. Pressure drop across biofiltration systems 1 and 2 was negligible when compared to values reported in literature for alternative filter bed media such as coconut fibre and peat. The pressure drop of biofiltration system 2 was 16 Pa greater than biofiltration system 1 at a maximum volumetric loading rate of $1847 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$.

When designing a biofiltration system for large scale operations on pig facilities, it is recommended that a maximum volumetric loading rate of $1350 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$ be used in order to achieve removal efficiencies greater than 90% for odour and ammonia emissions. This is calculated by multiplying $1693 \text{ m}^3 [\text{gas}] \text{ m}^{-3} [\text{medium}] \text{ h}^{-1}$ by 0.8 in order to implement a 20% safety factor suggested in Devanny et al. (1999). This equates to a filter size area of $0.148 \text{ m}^2 \text{ pig}^{-1}$ in summer conditions. This is quite similar to the recommended filter size areas of $0.125 \text{ m}^2 \text{ pig}^{-1}$ reported by Mannebeck (1994), but 41% and 55% respectively lower than the figures of 0.23 and $0.33 \text{ m}^2 \text{ pig}^{-1}$ reported by Scholtens et al. (1987). A biofilter medium chip size of greater than 20 mm using a 0.5 m bed height is recommended to reduce pressure drop and to

avoid the risk of developing anaerobic zones due to the high moisture content of the filter medium.

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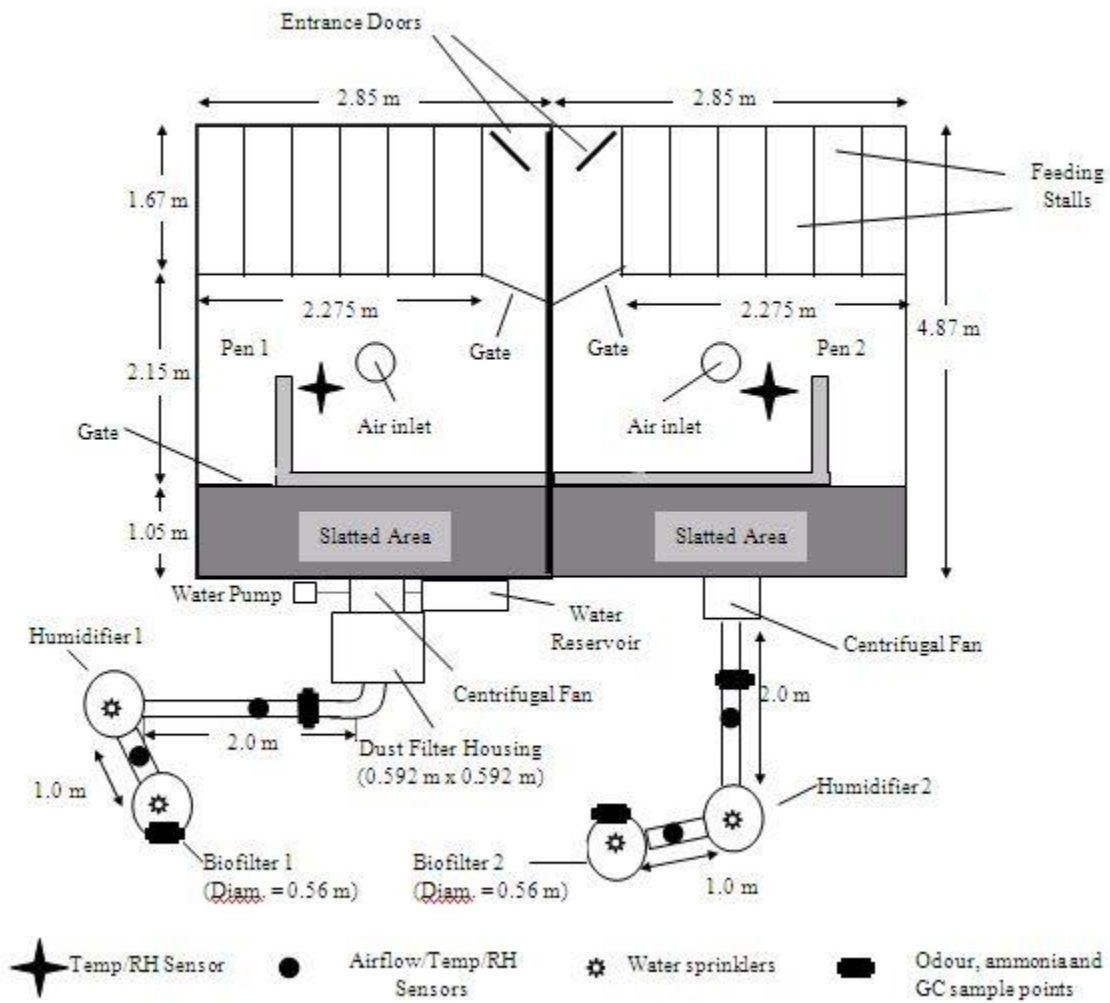


Fig. 1. Diagram of biofiltration systems (plan view).

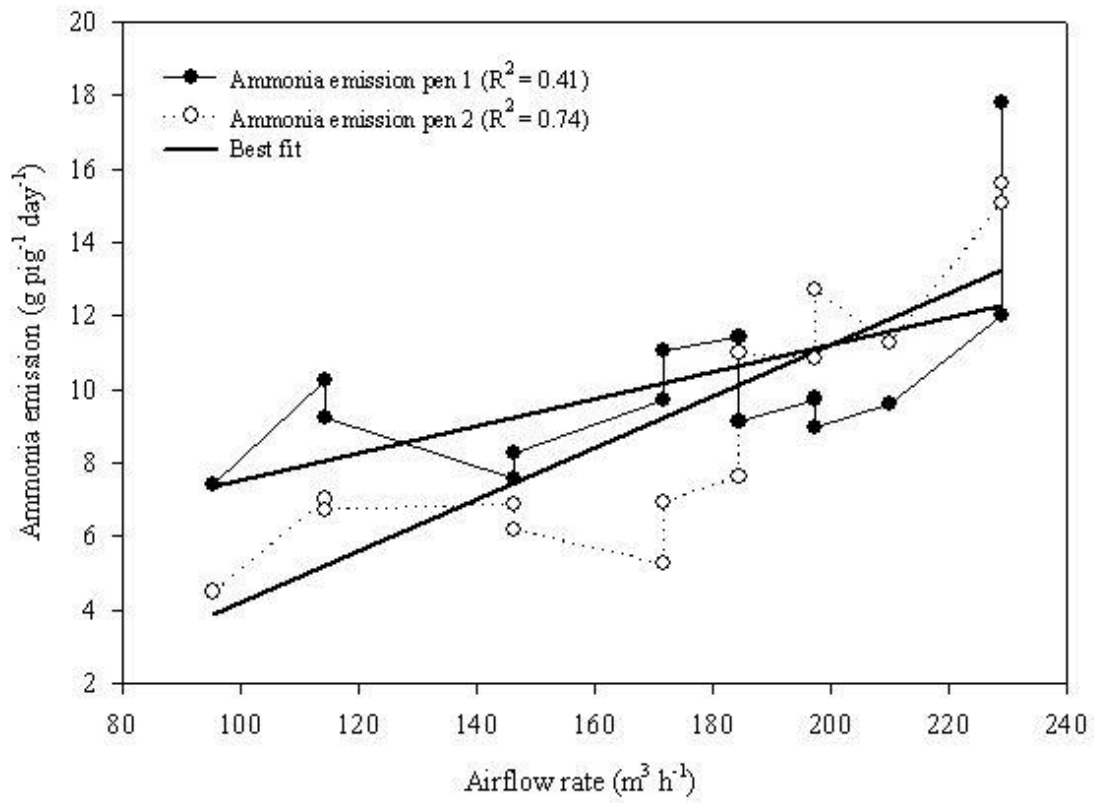


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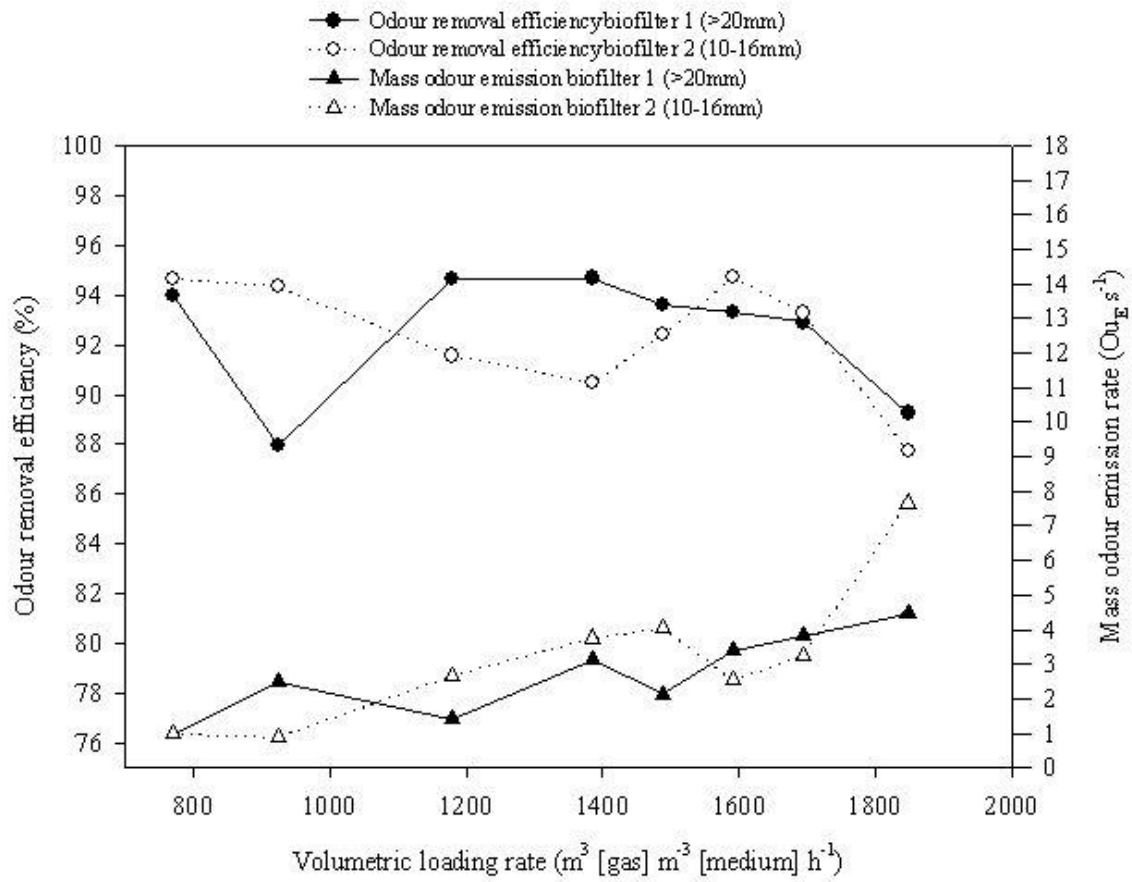


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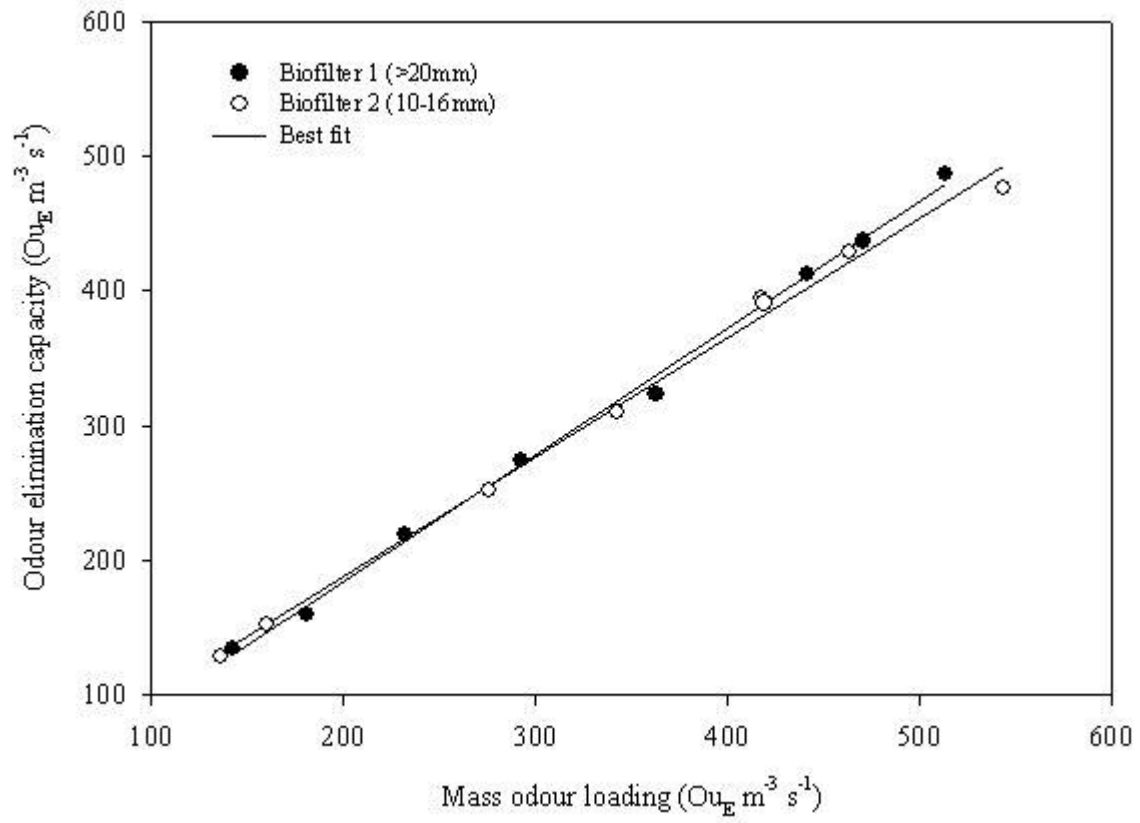


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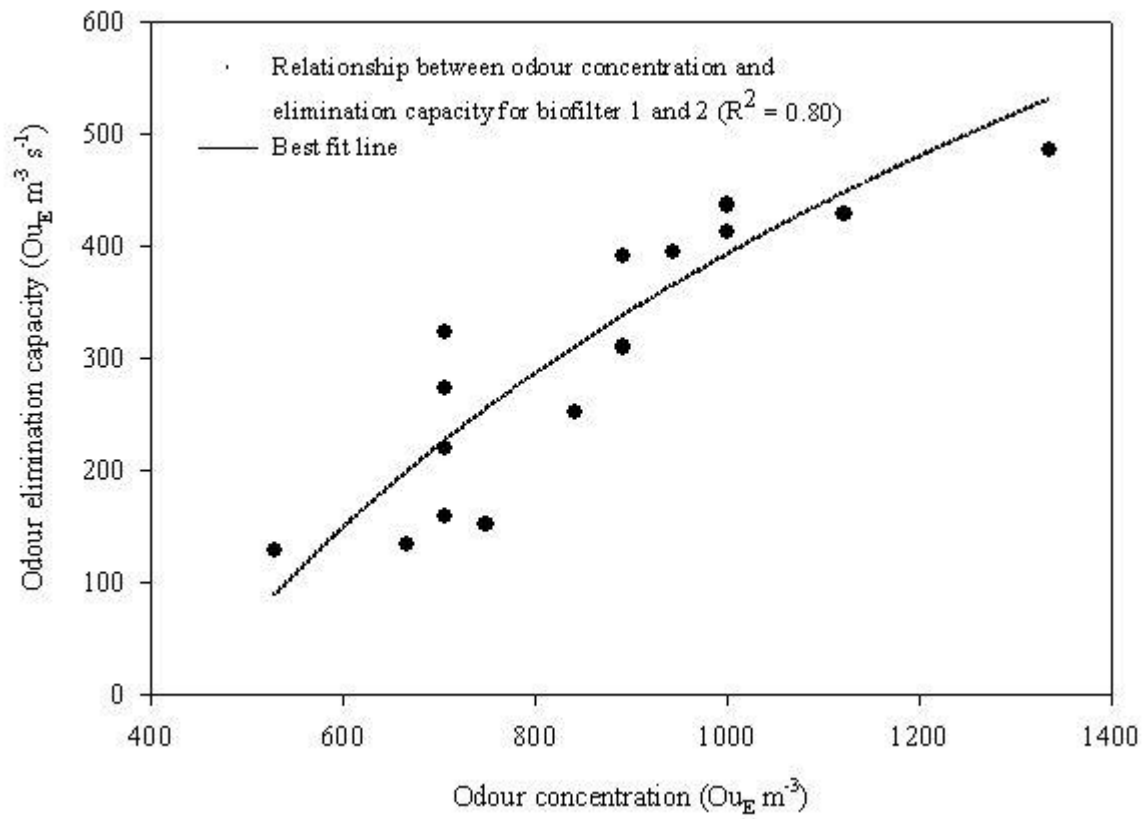


Fig. 5. Graph of odour elimination capacity of biofilters 1 and 2 versus odour concentration.

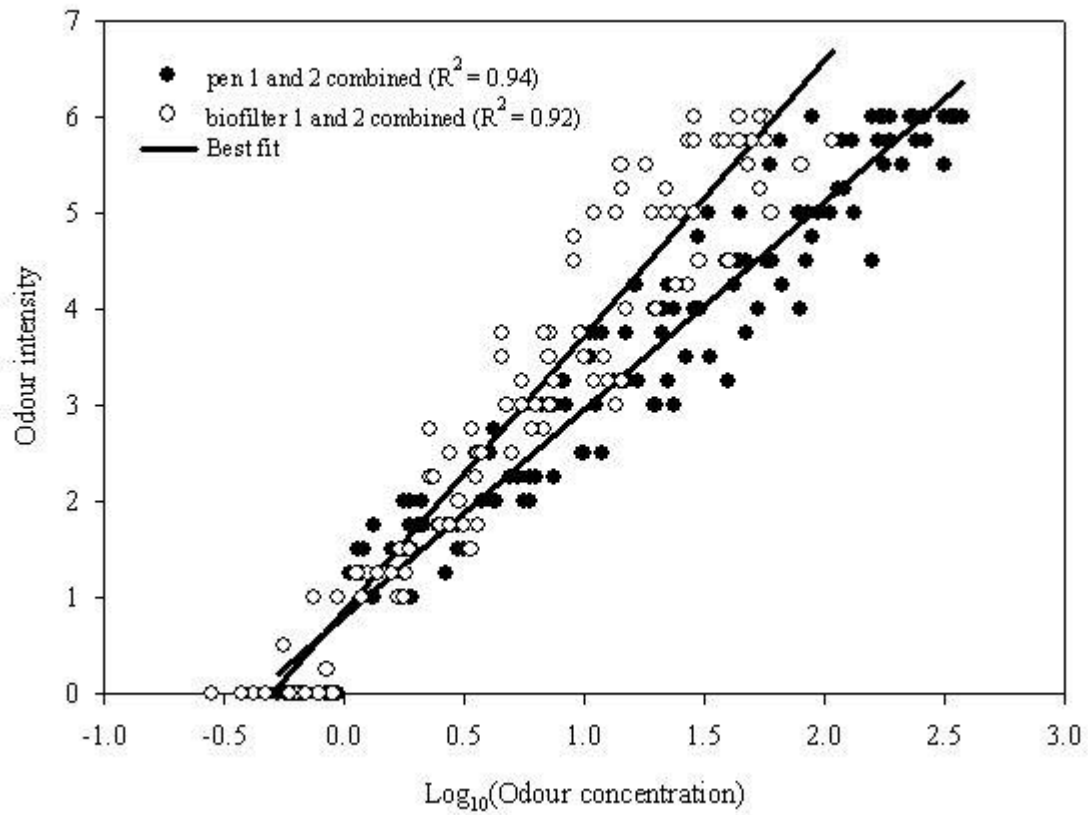


Fig. 6. Graph of odour intensity before and after biofilters 1 and 2 versus log₁₀ (odour concentration).

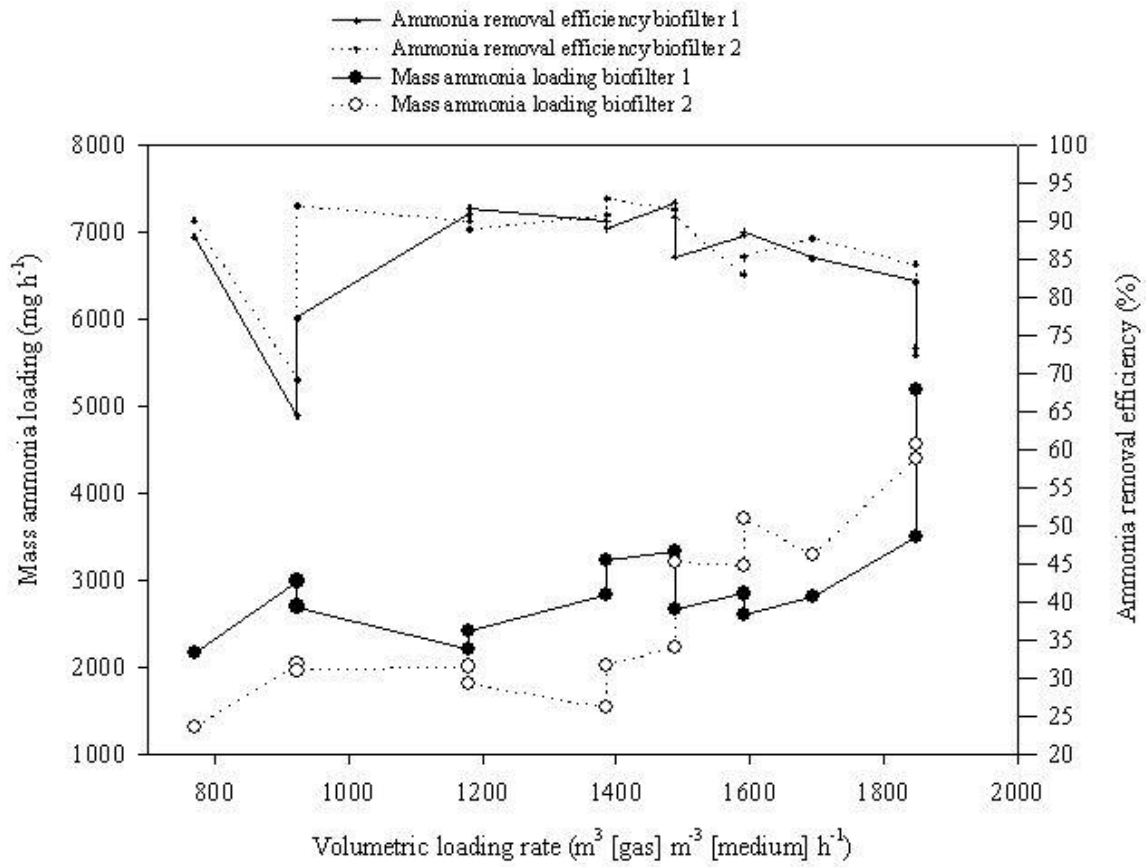


Fig. 7. Graph of mass ammonia loading and removal efficiency for biofilters 1 and 2 versus volumetric loading rate.

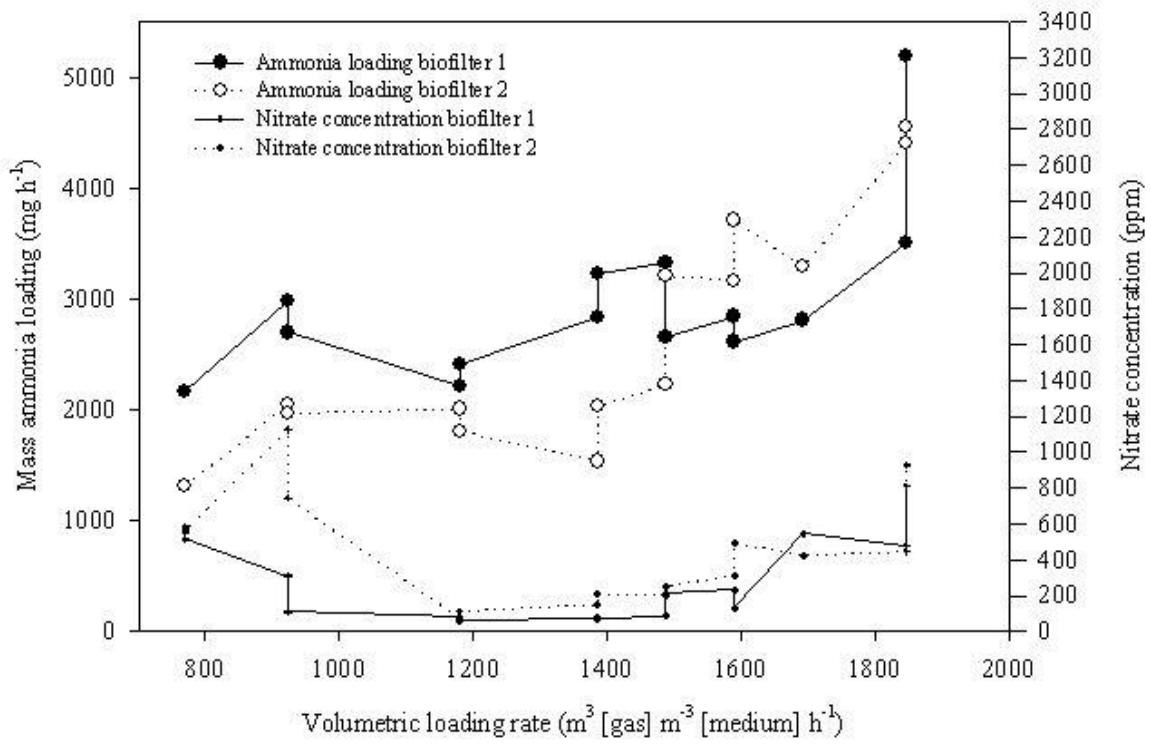


Fig. 8. Graph of mass ammonia loading and nitrate concentration for biofilters 1 and 2 versus volumetric loading rate.

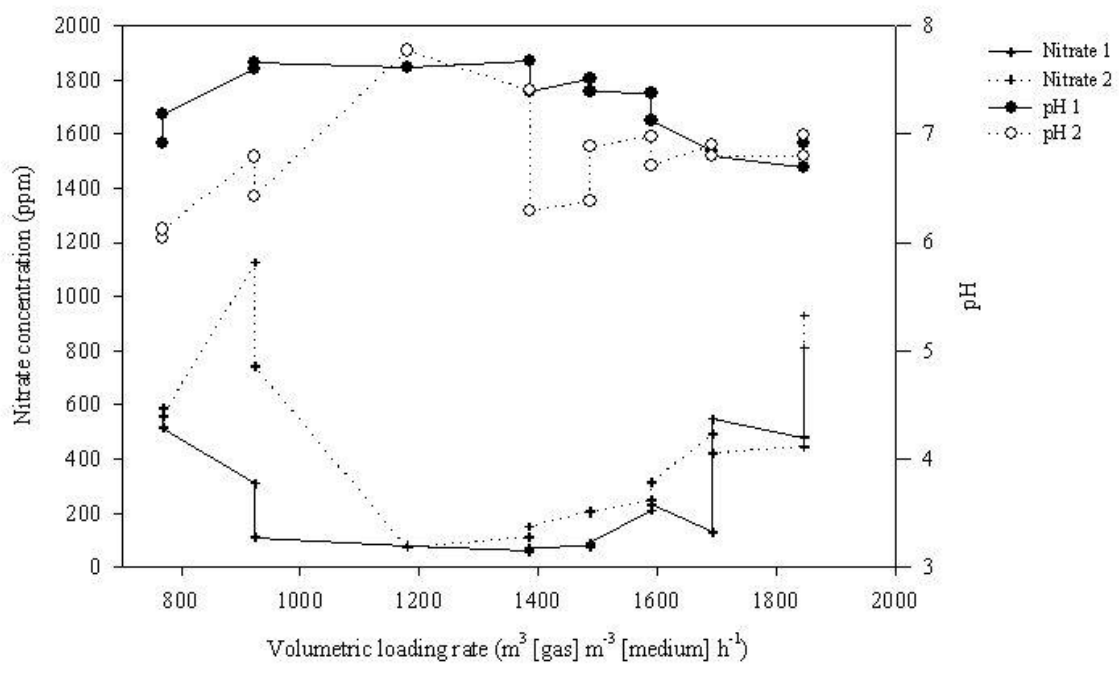


Fig. 9. Graph of nitrate concentration and pH of biofilters 1 and 2 versus volumetric loading rate.

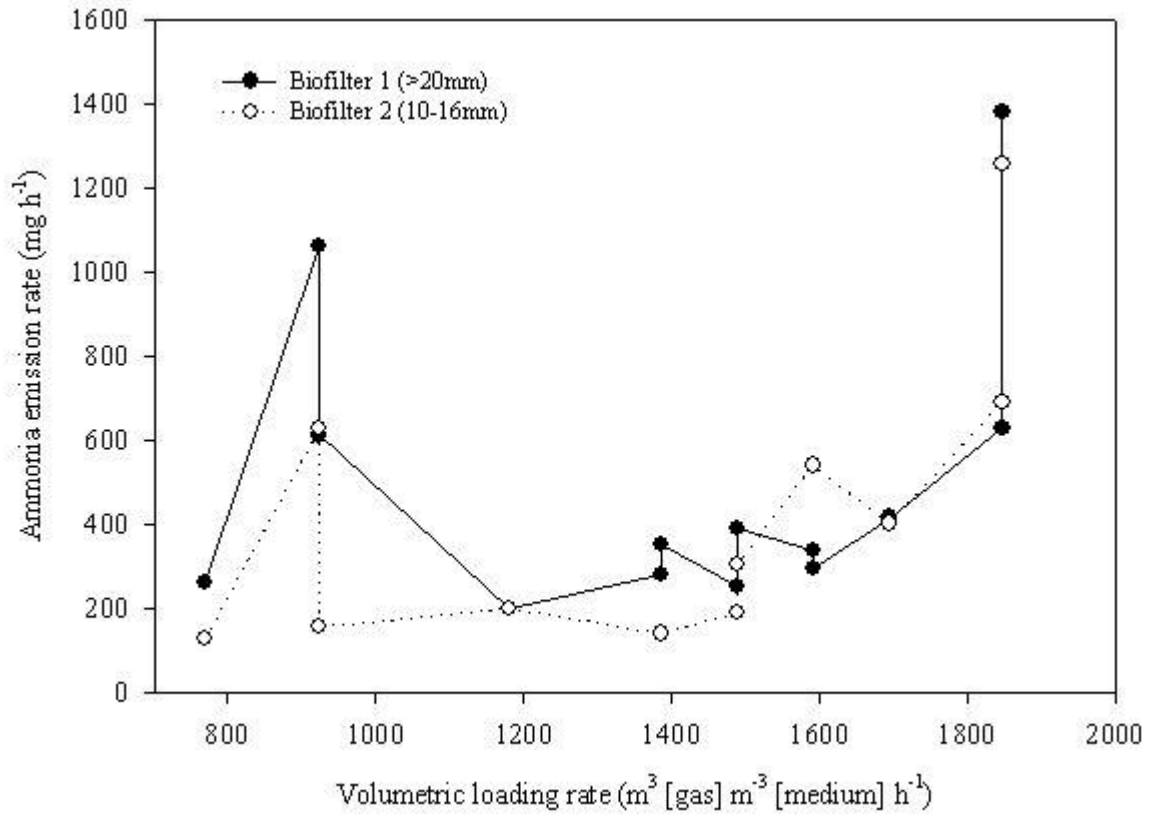


Fig. 10. Graph of ammonia emission rate from Biofilters 1 and 2 versus Volumetric loading rate.

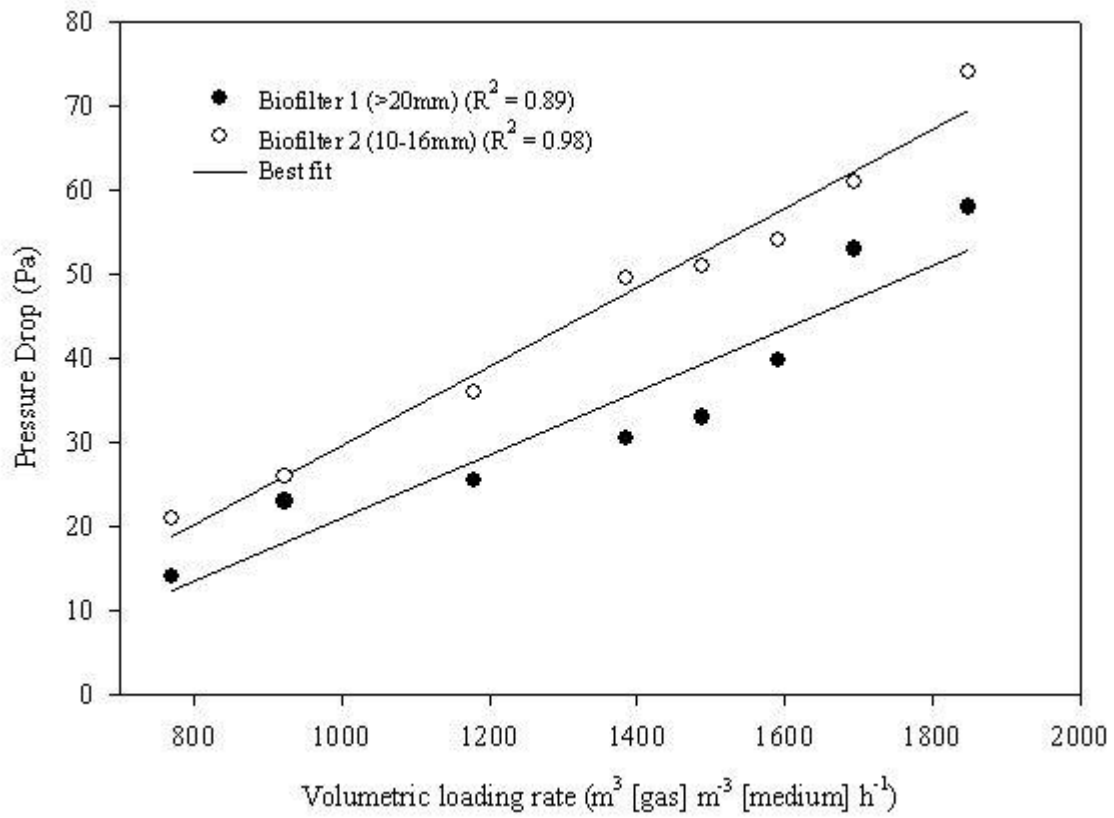


Fig. 11. Graph of pressure drop for biofilters 1 and 2 versus volumetric loading rate.

Table 1. Odour intensity relationship for pens 1, 2, exhaust biofilter 1 and exhaust biofilter 2.

Sample	Odour intensity relationship	R² correlation coefficient
Pen 1	$2.022(\text{Log}_{10} C) + 0.9178$	0.9363
Pen 2	$2.2834(\text{Log}_{10} C) + 0.6838$	0.9532
Exhaust biofiltration system 1	$2.7803(\text{Log}_{10} C) + 0.8267$	0.9239
Exhaust biofiltration system 2	$3.0176(\text{Log}_{10} C) + 0.8867$	0.9213

Table 2. Sulphur peak reduction for biofiltration systems 1 and 2 at various volumetric loading rates.

Volumetric loading rate (m³ [gas] m⁻³ [media] h⁻¹)	Sulphur peak reduction for biofiltration system 1 (%)	Sulphur peak reduction for biofiltration system 2 (%)
769	48	25
923	60	47
1180	53	-147
1385	65	-24
1488	60	50
1590	50	40
1693	60	46
1847	8	32