Development of an odorant emission model for sewage treatment works

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Abstract In the field of odour assessment, much attention has been paid to the measurement of odour concentration. Whilst the concentration of an odour at a receptor is a useful indicator of annoyance, the concentration at the source tells only half the story. The emission rate – the product of odour concentration and air flow rate – is required to appreciate the significance of odour sources. Knowledge of emission rates allows odour sources to be ranked in terms of significance and facilitates appropriate selection and design of odour control units. The emission rate is also a key input for atmospheric dispersion models. Given the increasing importance of odour to sewage treatment works operators, there is a clear need for predictive methods for odour emission rates. Theory suggests that the emission of odorants from sewage to air is controlled by mass transfer resistances in both the gas and liquid phase. These are in turn controlled by odorant and emission source characteristics. The required odorant characteristics are largely known, and mass transfer from many different types of emission sources have been studied. Sewage treatment processes can be described by one or more of six characteristic emission sources, these being quiescent surfaces, channels, weirs and drop structures, diffused aeration, surface aeration and flow over media. This paper describes the development of odorant mass transfer models for these characteristic emission types. The models have been applied in the form of spreadsheet models to the prediction of H₂S emissions and the results compared with commercial VOC emission models.

Keywords Emission; hydrogen sulphide; mass transfer; odour; sewage treatment

Introduction In the field of odour assessment, much attention has been paid to the measurement of odour concentration or strength (Gostelow et al., 2001). While the concentration of an odour at the receptor is a useful measure of potential for annoyance, the odour emission rate – the product of air flow rate and odour concentration – is required to fully appreciate the significance of odour sources. Knowledge of emission rates allows odour sources to be ranked in terms of significance and facilitates appropriate selection and design of odour control units. The emission rate is also a key input for dispersion models. Recent moves towards the use of source measurement and dispersion modelling to demonstrate compliance to nuisance standards (e.g. McIntyre, 2000, Yang and Hobson, 2000) means that knowledge of odour emission rates is becoming increasingly important.

The derivation of emission rates is straightforward where the odorous air is vented through a stack, but the majority of odour sources at sewage treatment works are large, open tanks with no well defined air flow (area sources). Special measurement techniques are required for area sources, involving enclosure of a portion of the emission surface or the use of atmospheric dispersion models. Alternatively, odorant concentrations can be measured in the liquid-phase and use mass-transfer models to predict emission rates.

Mass transfer theory Mass transfer of volatile compounds from water to air can be described by the equation:

\[ J = K_M (C_L - C_{L,e}) \]
where $J =$ specific emission rate ($g \, s^{-1} \, m^{-2}$), $K_M$ is the overall mass-transfer coefficient ($m \, s^{-1}$), $C_L =$ concentration of volatile compound in liquid-phase ($g \, m^{-3}$), $C_{L,e} = C_G / K_H$, $C_G =$ gas-phase odorant concentration ($g \, m^{-3}$), $K_H =$ Henry’s law constant (dimensionless).

The overall mass-transfer coefficient is made up of liquid and gas-phase local mass-transfer coefficients, $k_L$ and $k_G$ respectively. The overall mass-transfer coefficient is related to the local transfer coefficients by:

$$\frac{1}{K_M} = \frac{1}{k_L} + \frac{1}{K_H k_G}$$  \hspace{1cm} (2)

Equation (2) shows that the Henry’s law constant has a significant influence on whether gas or liquid-phase resistances are dominant. For highly volatile compounds, mass-transfer is governed by conditions in the liquid-phase. Conversely, for low-volatility compounds, gas-phase conditions are dominant. Most odorants are of moderate volatility and both gas and liquid-phase conditions are significant.

Emission models tend to utilise expressions for both $k_G$ and $k_L$ or they may relate the mass-transfer of a specific compound to the mass-transfer of oxygen, as oxygen transfer has been studied in some detail. In this case mass-transfer coefficients are related by (Munz and Roberts, 1989):

$$\frac{K_{L,C}}{K_{L,O}} = \left( \frac{D_{L,C}}{D_{L,O}} \right)^n + \frac{1}{K_H K_G / k_L}$$  \hspace{1cm} (3)

where $K_M =$ Overall mass-transfer for specific compound (C) or oxygen (O) ($m \, s^{-1}$), $D_L =$ Liquid-phase diffusion coefficient of specific compound (C) or oxygen (O) ($m^2 \, s^{-1}$).

Theoretical approaches have suggested that the exponent $n$ should be between 0.5 – 1.0 (Lewis and Whitman, 1924, Higbie, 1935, Danckwerts, 1951) whereas in practice most experimental work has indicated $n = 0.6$ (Corsi et al., 1992, Dewulf et al., 1998). The ratio $k_G / k_L$ which is dependent on the operating conditions of the unit process and is usually determined empirically.

**Emission models for unit processes**

Although there are a wide variety of unit processes at a sewage treatment works, emissions from most processes can be modelled using one or more of six emission situations, these being quiescent surfaces, channels, weirs or drop structures, diffused-air aerated tanks, surface aerated tanks, and flow over packed media (trickling filters). The development of emission models for these situations is described below.

**Emissions from quiescent surfaces**

Quiescent surfaces are common at sewage treatment works, with sedimentation tanks being an example. Emissions are largely controlled by the wind, affecting both gas-phase and liquid-phase mass-transfer coefficients through turbulence generation, wave formation and creation of drift currents (Komari et al., 1993).

It is important to appreciate the nature of the velocity profile close to the emission surface. This can be described by the logarithmic velocity distribution:

$$U_z = \frac{U^*}{k} \ln \frac{z}{z_0}$$  \hspace{1cm} (4)

where $U_z =$ Velocity at height $z$ ($m \, s^{-1}$), $U^*$ is the Friction velocity ($m \, s^{-1}$), $k$ is the von Karman constant (0.4), $z_0$ is the aerodynamic surface roughness height ($m$).

The friction or shear velocity $U^*$ is a useful correlating parameter, as this is effectively the wind velocity acting upon the water surface. It is, therefore, directly relevant to
turbulence conditions at the water-air interface.

A number of environmental friction velocity correlations have been produced (e.g. Wu, 1969, Smith, 1980, Wanninkhof and Bliven, 1991) which give similar results and correlate the friction velocity against the wind velocity measured at 10 m. For field measurements, a reference measurement of 2 m was thought to be more suitable, and a correlation based on 2 m wind velocity, based on the correlation of Smith (1980), is:

\[ U^* = 10^{-2}(5.26 + 1.36U_2)^{0.5}U_2 \]  

where \( U_2 \) is the wind velocity at 2 m (m s\(^{-1}\)).

Mass transfer coefficient correlations have been determined using wind-wave tunnel studies by several authors (Liss, 1973, Cohen et al., 1978, Wanninkhof and Bliven, 1991, Ocampo-Torres and Donelan, 1994). These studies were selected because they correlated both liquid and gas-phase mass-transfer coefficients against friction velocity for several gases and also because they provide a range of tank aspect ratios similar to that which may be seen in the field. The wind-wave tunnel studies have been generalised by the use of the Schmidt number, which represents the relative importance of the momentum and molecular diffusion components of mass-transfer and embodies temperature and compound-related influences on mass-transfer (Mackay and Yeun, 1983, Jahne and Haussecker, 1998). The Schmidt number is given by:

\[ Sc = \frac{m}{D} \]  

where \( m \) is the viscosity (Ns m\(^{-2}\)), \( r \) is the density (kg m\(^{-3}\)) and \( D \) is the diffusion coefficient (m\(^2\) s\(^{-1}\)).

Resulting mass-transfer correlations from the above mentioned wind-wave tunnel studies are:

\[ k_L = 0.0035U \ Sc_L^{-0.5} \]  
\[ k_G = 0.04U \ Sc_G^{-0.67} \]  

**Emissions from channels**

Open channels are commonly used for conveyance of sewage at treatment works. There are relatively few models of emissions from gravity sewers or open channels. Those that exist tend to be based on empirical models for oxygen absorption.

Experiments and reviews of previous work by Balmer and Tagizadeh-Nasser (1995) indicated that energy dissipation and hydraulic mean depth are the most significant correlating factors for oxygen mass-transfer. Their experiments, using a 24 m sewer section with adjustable slope, indicated little justification for inclusion of the Froude number as used in the Parkhurst and Pomeroy (1972) model which is in common use. Balmer and Tagizadeh-Nasser (1995) produced the following correlation:

\[ K_{M,O} = 1.06 \ 10^{-3} E^{0.375} d_m^{0.4} \text{ m s}^{-1} \]  

where \( K_{M,O} \) is the overall mass-transfer coefficient for oxygen (m h\(^{-1}\)), \( E \) is the energy dissipation with \( E = Us_t g \), where \( s \) is the dimensionless channel slope and \( g \) is the acceleration due to gravity (9.81 m s\(^{-2}\)); \( d_m \) is the hydraulic mean depth (m).

This was very similar to earlier work performed by Krenkel and Orlob (1962) using laboratory flumes and appears to be a good approximation for oxygen mass-transfer from channels.

**Emissions from weirs and drop structures**

Weirs and drop structures are difficult to model due to the number of emission mechanisms occurring. Models for weirs and drop structures (Nakasone, 1987, Pincince, 1991, Labocha
et al., 1996, Zytner et al., 1997, Rahme et al., 1997, Zytner et al., 1999) tend to utilise the deficit ratio which is defined as:

\[ r = \frac{C_u}{C_d} \frac{C_G}{K_H} / \frac{C_G}{K_H} \]  

where \( r \) is the deficit ratio, \( C \) is the dissolved gas concentration upstream (u) or downstream (d) of the weir.

With knowledge of the deficit ratio, the emission rate from the weir can be determined from the product of the difference in upstream and downstream concentrations and the liquid flow rate. Typically, deficit ratios have been derived for oxygen. The oxygen deficit ratio can be related to an odorant emission rate using:

\[ r_c = r_o \]  

where \( r_c \) is the deficit ratio for specific compound, \( r_o \) is the deficit ratio for oxygen, and the proportionality factor \( y = K_{L,C}/K_{L,O} \). The proportionality factor is given by Equation (3).

Experimental correlations (Nakasone, 1987, Pincince, 1991, Labocha et al., 1996, Rahme et al., 1997) for deficit ratios have shown weir loading rate (or flow rate for pipe discharges) and drop height to be the most significant factors. Wastewater quality was also found to be a significant factor. Consideration of these experiments has produced the following correlation for weir emissions:

\[ r = \exp(0.2585Z^{0.023}q^{-0.463}) \]  

where \( Z \) is the drop height (m), \( q \) is the weir loading (m\(^3\)/m\(^-1\)/h\(^-1\)), is the wastewater quality parameter, equal to 0.4 for primary wastewater and 1 for secondary wastewater.

For drop structures involving pipe discharges, the correlation of Rahme et al. (1997) is used:

\[ \ln r = 0.731Z^{0.784}Q^{-0.139} \]  

where \( Q \) is the wastewater flow from pipe (m\(^3\)/h\(^-1\)).

\( K_G/K_L \) ratios for weirs and drop structures have been measured by Rahme et al. (1997) and Zytner et al. (1999). These were in the range of 1.3–2.2 and were found to be insensitive to weir operating conditions.

**Emissions from diffused-aeration processes**

For bubble-aerated tanks, there is a well defined air flow through the tank. Emission rates can be determined from the odorant concentration in the air bubbles leaving the tank surface multiplied by the air flow.

Several models have been developed for emissions from aeration tanks (Chern and Yu, 1995; Liao and Lee 1996, 1997a, 1997b), but the Matter-Muller et al. (1981) mass-balance model has the advantage of simplicity and is applicable for most situations. The model utilises the \( K_{L,a} \) value for oxygen, which is a known or easily derived parameter for most tanks. The emission rate determined by the Matter-Muller et al. (1981) model is:

\[ M = Q_G C_L K_H \exp\left(\frac{-K_{L,a}V_L}{K_H Q_G}\right) \]  

where \( M \) is the emission rate (g s\(^-1\)), \( Q_G \) is the gas flow rate (m\(^3\)/s\(^-1\)) and \( V_L \) is the liquid volume (m\(^3\)).

In order to use Equation (13) for compounds other than oxygen, the appropriate \( K_{L,a} \) value should be substituted, which can be determined using equation (3). Appropriate \( K_G/K_L \) ratios appear to be in the range of 2.2–3.6 according to experiments by Hsieh et al. (1994).
Emissions from surface-aerated processes

Experimental work using laboratory-scale reactors (Roberts and Dandliker, 1983; Munz and Robert, 1989) has indicated that for surface aerated tanks the oxygen mass-transfer coefficient is almost linearly proportional to aerator specific power input. This is reflected in a model often applied to full-scale reactors (Roberts et al., 1984):

\[ K_{L,a} = \frac{N \cdot P}{C_{O_2}^s \cdot V} \]  

(15)

where \( K_{L,a} \) is the oxygen mass-transfer coefficient (h\(^{-1}\)), \( N \) is the aerator standard oxygen transfer rate (kg O\(_2\) kW\(^{-1}\) h\(^{-1}\)), \( C_{O_2}^s \) is the oxygen saturation concentration (kg O\(_2\) m\(^{-3}\)), \( P \) is the aerator power (kW), \( V \) is the tank volume (m\(^3\)).

If efficiency, water quality and temperature parameters are added as per Thibodeaux (1996), and the oxygen saturation concentration at 20°C of 9.08 \( \times 10^{-3} \) kg m\(^{-3}\) (Metcalf and Eddy Inc., 1991) is included, the expression becomes:

\[ K_{L,a} = \frac{N \cdot P}{9.08 \times 10^{-3} \cdot V} (1.024)^{(T - 20)} \text{h}^{-1} \]  

(16)

where \( e \) is the aerator power delivery efficiency (dimensionless), \( \alpha \) is the dirty/clean water \( K_L \) ratio, and \( T \) is the temperature (°C).

\( K_G/K_L \) ratios for surface aeration have been investigated by several authors, (e.g. Roberts and Dandliker, 1983; Munz and Roberts, 1989; Hsieh et al., 1993; Parker et al., 1996) and appear to be in the range 16–120. The field-scale investigations of Parker et al. (1996) gave the lowest \( k_G/k_L \) ratios and were also measured over the lowest \( P/V \) range. A value of 30 is recommended.

Emissions from flow over media (trickling filters)

The most common application of flow over media in wastewater treatment is the trickling filter. For trickling filters, the odorant concentration in the liquid-phase changes through the depth of the filter through both biodegradation and volatilisation and both mechanisms must be included in an emission model. Melcer et al. (1995) and Parker et al. (1996) describe a trickling filter model incorporating these mechanisms which is based on mass-balances for horizontal slices of the filter bed.

To apply the Melcer et al. (1995) trickling filter model, expressions for the oxygen mass-transfer coefficient and the air flow through the filter bed and required. A correlation for \( K_{L,v,O} \) for nitrifying trickling filters was determined by Wik (1996), and is:

\[ K_{L,v,O} = 0.0120 q_L + 0.08541 \text{ (m} d^{-1}) \]  

(17)

where \( q_L \) is the hydraulic load (m\(^3\) m\(^2\) d\(^{-1}\)).

Air flow is governed by the temperature difference between air and wastewater and the head loss through the filter bed. The pressure head due to temperature difference is estimated using:

\[ \frac{P}{Z} = 3462 \left( \frac{1}{T_c} - \frac{1}{T_h} \right) \]  

(18)

where \( P \) is the pressure (Pa), \( T_a \) is the air temperature (K), \( T_w \) is the wastewater temperature (K).

The head loss through a trickling filter is estimated using the Ergun equation:

\[ \frac{P}{Z} = 150 \left[ \frac{(1 - \phi)^2 U_0}{d_p} + 1.75(1 - \phi)^2 \frac{U_0^2}{d_p^2} \right] \]  

(19)

where \( \phi \) is the air viscosity (kg m\(^{-1}\) s\(^{-1}\)), \( \rho \) is the air density (kg m\(^{-3}\)), \( \phi \) is the bed porosity, \( \phi \) is
the sphericity of media, \(d_p\) is the diameter of media (m), \(U_0\) is the superficial velocity (m s\(^{-1}\)).

The superficial velocity can then be found by trial and error using Equations (18) and (19) and the air flow is determined from the product of superficial velocity and filter plan area.

Application of the emission models

The emission models above have been applied in the form of spreadsheet models to the prediction of H\(_2\)S emissions from sedimentation tanks, channels, diffused-air and surface aeration and trickling filters. Results have been compared with commercial VOC emission models WATER8 and Toxchem+, available from USEPA and Enviromega, respectively. In each case, a liquid-phase free H\(_2\)S concentration of 1 g m\(^{-3}\) has been used and biodegradation has been ignored by setting relevant rate constants to zero. Selected comparisons are shown in Figure 1. In general, the spreadsheet models are in good agreement with Toxchem+ and WATER8 although some differences in the models are evident, particularly for quiescent surface and channel emissions.

For quiescent surfaces emissions, the spreadsheet model and Toxchem+ show a greater dependence on wind velocity than does WATER8. For short tanks, such as sedimentation tanks, WATER8 assumes that the effect of the wind on the liquid-phase mass-transfer is negligible for short tanks which seems to contradict many wind-wave tunnel studies. It is believed that the wind dependence shown by the spreadsheet model and Toxchem+ is more realistic.

Figure 1 Comparison of spreadsheet and WATER8 models. (a) quiescent surface; (b) weir; (c) channel; (d) diffused-air aeration; (e) surface aeration; (f) trickling filter
Channel emissions show significant differences, with the spreadsheet predicting a higher emission rate than WATER8 and Toxchem+. WATER8 and Toxchem+ show little variation with wastewater flow rate, but are affected by wind velocity. The spreadsheet model predicts emissions are driven largely by turbulence induced from the wastewater flow and does not include any wind effects. Due to a lack of published data, the spreadsheet model uses an assumed $K_G/K_L$ ratio of 100. In practice, a lower ratio may be appropriate which will reduce emissions as gas-phase resistance becomes more significant.

**Conclusions**

The importance of emission rates in odour assessments, combined with the difficulty of their measurement for many sewage treatment sources indicates that emission modelling, based on liquid-phase odorant concentrations is a potentially useful technique for estimating emission rates. Emission models for odorants have been developed and applied to H$_2$S emissions. Comparison with existing VOC emission software (WATER8 and Toxchem+) indicates that relatively simple spreadsheet models can be used to predict odorant emission rates. Where significant differences exist between WATER8, Toxchem+ and the spreadsheet models, it is believed that the spreadsheet models are more representative of true conditions.

**Recommendations for further work**

The spreadsheet models have at present received limited validation against other modelling packages. Further validation is planned against measured emission rates using hood and micrometeorological measurements for field-scale unit processes.

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**References**


