

## Reaeration in sewers

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### Abstract

The sewer system is a very dynamic system with an abundance of mass transfer processes and transformations. A key process is the mass exchange between the wastewater and the sewer atmosphere. An equation that describes the gas–liquid mass transfer under different hydrodynamic conditions is essential when sewer processes are to be quantified or modelled.

In this work, a calibrated reaeration equation is proposed. It is based on the shear Reynolds and the Froude number to correct the increased gas–liquid interface roughness to higher flow rates. The equation was calibrated with previously published data and with new data. This data was obtained with a safe and environmentally friendly gas tracer method for gravity sewers based on the inert gas sulphur hexafluoride (SF<sub>6</sub>), a new method for the sewer system. Measurements were conducted in four channels under different conditions. The resulting equation will allow for more accurate simulations of the sewer system. Finally, the effect of reaeration with regard to the oxygen consuming processes for different hydrodynamic conditions is discussed.

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### 1. Introduction

A key process in sewers and other open channels is the gas–liquid mass exchange between flowing water and the atmosphere. Compounds such as hydrogen sulphide or organic acids can be released, and oxygen can be taken up by the wastewater. This latter process, which is commonly known as reaeration, strongly influences the dissolved oxygen (DO) concentration of the wastewater. This oxygen is consumed during transformation processes that are mostly intermediated by bacteria in the wastewater and in the sewer wall biofilm. Especially, the

biofilm activity strongly depends on the DO concentration in the bulk liquid [1].

Aerobic conditions will prevail in the sewer when reaeration balances the microbial oxygen uptake. As a result, easily degradable organic matter ( $c_S$ ) will be completely oxidised to carbon dioxide and the wastewater will contain a low concentration of  $c_S$  when it reaches the wastewater treatment plant (WWTP). This will impair the nutrient removal capacity of WWTP. In contrast, the wastewater will become anaerobic when reaeration is smaller than oxygen uptake. Fermentation will prevail and the concentration of  $c_S$  can be expected to rise. Anaerobic conditions also promote the production of hydrogen sulphide, which leads to concrete erosion and causes unpleasant odours. Though even under anaerobic conditions, the conversion of organic compounds by microbiological activity will be related to the transfer of oxygen into the wastewater.

The reaeration should be known accurately when sewer processes are to be quantified, modelled or simulated. The sewer system is comparable to streams

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## 2. Physical background of gas–liquid mass transfer

More than 99.8% of the oxygen transfer resistance lies on the liquid side [15]. Therefore, the oxygen mass transfer rate,  $r_{O_2,m}$  can be written as

$$r_{O_2,m} = k_1 a (c_1^* - c_{b,l}), \quad (1)$$

where  $k_1 a$  is the reaeration coefficient with  $k_1$  being the mass transfer coefficient related to the liquid-phase,  $a$  the specific exchange surface area =  $w/A_{cr} = 1/d_m$ ,  $w$  the surface width,  $A_{cr}$  the cross sectional area,  $d_m$  the mean hydraulic depth,  $c_1^*$  the equilibrium concentration of solute in liquid phase,  $c_{b,l}$  the liquid-phase bulk concentration. The calculation scheme for the hydrodynamic variables ( $d_m$  etc.) is given in Appendix A.

In the case of wastewater systems, three factors interfere with the gas–liquid mass transfer. Firstly, very small amounts of impurities in the water (surfactants, fats and oils), which concentrate on the interfacial area can have a measurable effect on the oxygen mass transfer [16]. These substances occupy surface area, hampering the exchange of liquid elements at the surface [17]. This can be expressed by introducing the  $\alpha_{k_1 a}$ -factor, which is the ratio of  $k_1 a$  under practical conditions to that in clean water ( $k_1 a_{wastewater}/k_1 a_{clean\ water}$ ). Values for  $\alpha_{k_1 a}$  as low as 0.4 have been reported [8].  $\alpha_{k_1 a}$  depends on the wastewater characteristics that change in a geographical and diurnal manner and should therefore be quantified experimentally whenever possible.

Secondly, the saturation concentration of oxygen in wastewater is reduced due to water vapour in the sewer atmosphere, dissolved salts in the water and a reduced oxygen fraction in the sewer atmosphere due to the uptake by the wastewater [18]. These three influences can be lumped in a factor  $\beta$ , which is the ratio of solubility under practical conditions to that in clean water. Values of 0.95 [10] and from 0.97 to unity [19] have been published. Normally, sewers are well aerated to prevent to build up of toxic gases. The water vapour in the sewer atmosphere will easily reach saturation because the wastewater is normally warmer than the sewer atmosphere and the sewer walls. The partial oxygen pressure, after correction for partial vapour pressure, is therefore practically equal to the outside atmosphere, as was also found by Parkhurst and Pomeroy [20]. Eventhough this factor could be determined experimentally also, an estimate based on the corrected oxygen partial pressure (water saturated sewer atmosphere and height above sea level) and an average salt strength is normally sufficient because the error introduced this way (<5%) is marginal in comparison to other errors.

Eq. (1) can now be rewritten as

$$r_{O_2,m} = \alpha_{k_1 a} k_1 a (\beta \cdot c_1^* - c_{b,l}). \quad (2)$$

### 2.1. Reaeration coefficient

The gas–liquid mass transfer can be described with the dimensionless Sherwood number ( $Sh$ ) as a function of the Reynolds ( $Re$ ) and Schmidt ( $Sc$ ) numbers (after Bennett and Myers [21])

$$Sh \propto Re^p \cdot Sc^q. \quad (3)$$

The exponent  $p$  is commonly taken as  $\frac{1}{2}$  [21,22]. The value of  $Sc$  for a given solute/solvent system is only affected by the temperature. However, its exponent  $q$  depends on the turbulence level in the water phase at the gas–liquid interface [23]. This exponent can be written as  $(1 - n)$  where the constant  $n$  depends on the interfacial turbulence as described by a boundary layer model. There are three basic models [23]: the film-theory when  $n = 1$  [24], the penetration theory when  $n = \frac{2}{3}$  [25] and the surface renewal theory when  $n = \frac{1}{2}$  [26]. With the definitions of the dimensionless numbers included, (3) can be transferred into

$$\frac{k_1 L}{\mathcal{D}} \propto \left(\frac{uL}{\nu}\right)^{1/2} \left(\frac{\nu}{\mathcal{D}}\right)^{1-n}, \quad (4)$$

where  $L$  is the characteristic length,  $\mathcal{D}$  the diffusion coefficient,  $\nu$  the kinematic viscosity =  $\eta/\rho$ ,  $\eta$  the dynamic viscosity and  $\rho$  the density.

Eq. (4) is useful for two aspects of gas–liquid mass transfer

#### 2.1.1. Temperature conversion

The temperature dependency of the gas–liquid mass transfer is normally described with the following empirical equation [27]:

$$\frac{k_{1,T}}{k_{1,T_{ref}}} [\text{Oxygen}] = \theta^{(T-T_{ref})} = 1.0241^{(T-20)}, \quad (5)$$

where  $\theta$  is the exponential temperature coefficient,  $T$  the temperature and  $T_{ref}$  the reference temperature (normally 10°C cor 20°C). Although this equation is only valid for oxygen, it has been used for other gases in the past (e.g. by Jensen and Hvitved-Jacobsen, [28]). Furthermore, this equation does not take the boundary layer conditions into account. In contrast, with the quotient of (4) for  $T = T$  and for  $T = T_{ref}$  a general temperature dependency can be defined:

$$\frac{k_{1,T}}{k_{1,T_{ref}}} [\text{General}] = \left(\frac{\mathcal{D}_T}{\mathcal{D}_{T_{ref}}}\right)^n \left(\frac{\nu_T}{\nu_{T_{ref}}}\right)^{1/2-n}. \quad (6)$$

The conversion with this equation becomes equal to that of (5) when  $n = 0.72$  is applied. Consequently, (5) is specific for oxygen and a boundary layer model with characteristics somewhere between film and surface renewal theory. In contrast, (6) can be used for any compounds by inserting the compound-specific and readily available diffusion coefficients.

### 2.1.2. Conversion between $k_1$ of different compounds

Eq. (4) is useful to transfer the  $k_1$  of one compound, e.g. a gas tracer, into that of another, e.g. oxygen. This conversion follows from the quotient of (4) for both compounds, which results in

$$\frac{k_{1,\text{tracer}}}{k_{1,\text{O}_2}} = \left( \frac{\mathcal{D}_{\text{tracer}}}{\mathcal{D}_{\text{O}_2}} \right)^n \quad (7)$$

The outcome of the conversions with (6) and (7) depends on the boundary layer model. It is therefore advisable to determine which model is valid for a certain system. This has been done experimentally for this work as will be described further down below.

### 2.2. Reaeration equations

Several researchers have proposed equations for oxygen mass transfer in smooth channels or sewers (If necessary  $k_1$  was converted to  $k_1a$  by dividing by  $d_m$ ) (at 20°C) (see Table 1).

Moog and Jirka [7] compared 11 equations for river reaeration and showed that the channel slope, the water depth and the water velocity are the major variables. Furthermore, these authors discussed that turbulent flow is characterised by eddies, which will strongly affect the gas–liquid mass transfer. From the surface renewal model [26], a general equation for the mass transfer was deduced (after Moog and Jirka [13]):

$$k_1a \propto \frac{u^*(Re^*)^m Sc^{-1/2}}{d_m} \quad (18)$$

where  $Re^*$  is the shear Reynolds number (dimensionless):

$$Re^* = \frac{u^* d_m}{\nu} \quad (19)$$

and  $u^*$  is the shear velocity:

$$u^* = \sqrt{gR_h S_f} \quad (20)$$

$g$  is the gravitational acceleration,  $R_h$  the hydraulic radius and  $S_f$  the friction slope.

The exponent of  $Re^*$  depends on the prevailing eddy-model and therefore has a different physical meaning than the Reynolds number in (3). On theoretical grounds,  $m = -\frac{1}{2}$  was deduced for the large eddy model where the eddy-size is presumed to be proportional to the flow depth [13,30]. When small eddies in the range of the Kolmogorov length are prevailing, then  $m = -\frac{1}{4}$  [31].

Eq. (18) is based on sound theoretical considerations. However, in order to describe the gas–liquid mass transfer under a wide range of conditions, the positive effect of an increased roughness of the gas–liquid interface at supercritical flow should be added. The dimensionless Froude number ( $Fr$ ) is commonly used to quantify the effect of supercritical flow in fluid motion:

$$Fr = \frac{\bar{u}^2}{gd_m} \quad (21)$$

The effect will be demonstrated later with the experimental results. The equation for the reaeration coefficient becomes:

$$k_1a = a_{k_1a} \frac{u^*(Re^*)^m Sc^{-1/2}(1 + b_{k_1a} Fr)}{d_m}, \quad (22)$$

where  $a_{k_1a}$  and  $b_{k_1a}$  are constants. Eq. (22) will be used to fit the data acquired during this study and previously published data.

### 2.3. Reaeration measurement in the sewer system

Two different methods have been applied in the past to quantify sewer reaeration.

- The *indirect method* is based on an oxygen mass balance [8]. All terms of the oxygen balance are either measured or estimated. For application in the sewer, it requires first a mechanical cleansing of the sewer wall followed by huge amounts of aggressive chemicals to eliminate the loss of oxygen through respiration.
- *Direct methods* use stable chemicals with properties similar to oxygen as a tracer to determine the reaeration rate. Normally, this compound is added

Table 1

Type	$k_1a(\text{h}^{-1})$	Reference	Eq. No.
A	$7.235(S_0u)^{0.408} d_m^{-2/3}$	Krenkel and Orlob [11]	(8)
	$0.222u^{0.67} d_m^{-1.85}$	Owens et al. [6]	(9)
	$2.10(S_0ug)^{3/8} (d_m/D)^{0.4} d_m^{-1}$	Balmér and Tagizadeh-Nasser [12]	(10)
	$0.161Sc^{-0.5} [(u^*v)/d_m]^{1/4} d_m^{-1}$	Moog and Jirka [13]	(11)
B	$0.45(1 + Fr^{1/4}) \sqrt{(d_m g S_0)} d_m^{-1}$	Thackston and Krenkel [4]	(12)
	$0.96(1 + 0.17Fr)(S_0u)^{3/8} d_m^{-1}$	Parkhurst and Pomeroy [20]	(13)
	$720uS_0$	Tsivoglou and Neal [29]	(14)
	$0.4u(d_m/R_h)^{0.613} d_m^{-1}$	Taghizadeh-Nasser <sup>a</sup>	(15)
	$0.96(1 + 0.17Fr)(S_0u)^{0.75} d_m$	Jensen and Hvitved-Jacobsen [28]	(16)
	$0.86(1 + 0.20Fr)(S_0u)^{3/8} d_m^{-1}$	Jensen [10]	(17)

<sup>a</sup> in Jensen [10]. A is the simulated stream or sewer and B the real open channel. See the Nomenclature for the definition of variables.

to the water phase and its decrease over a channel section is quantified. The best known direct method is based on the radioactive gas krypton-85 ( $^{85}\text{Kr}$ ) as a tracer gas [28]. However, with regard to effects on the environment, authorities and personnel, it is undesirable to use radioactive tracers.

An alternative, easy, environmentally friendly, safe and accurate method for the sewer system was needed. Therefore, a method based on the inert gas sulphur hexafluoride ( $\text{SF}_6$ ) was adapted for the sewer system. sulphur hexafluoride has many desirable features as a deliberate tracer, most notably its low detection limit ( $\approx 50 \times 10^{-12} \text{ g L}^{-1}$ ), its long term stability in water, the possibility to preserve water samples containing  $\text{SF}_6$ , and ease of analysis [32].  $\text{SF}_6$  has been used as a tracer in lakes and rivers (e.g. [14,32]) but the confined structure, the wastewater and the thick, fat-containing biofilm mean that care should be taken when this method is applied in the sewer system. It must be confirmed that the major drawback of  $\text{SF}_6$ —its high octane–water partition coefficient—does not interfere with the gas exchange.

### 3. Materials and methods

#### 3.1. Tracer preparation and addition

The  $\text{SF}_6$ -tracer could either be added as a pulse, or as a step. In rivers, pulse addition is almost always used due to the large flows that would otherwise require a large quantity of tracer. However, pulse addition requires a non-volatile dispersion tracer (radioactive  $^3\text{H}$  (tritium) [10,28,33], rhodamine, sodium chloride or sodium bromide) because the tracer peak will decrease and widen due to dispersion. The ratio between the dispersion and gas tracers concentration at each site is used to calculate the gas tracer mass transfer coefficient e.g. [10]. However, the additional tracer increases the analytical burden and will reduce the accuracy of the mass transfer quantification.

In the sewer system, the dispersion is normally much smaller than in a river. In the case of pulse addition, very fast sampling would be required to resolve the tracer peak. The extra equipment that would be needed would introduce an extra error and complicate field experiments (see e.g. [28]). Fortunately, the discharge in sewers is generally smaller than in rivers. Therefore, continuous addition (a step) of a gas tracer is possible and samples can be taken over a prolonged period. This is the method that has been used during this study. But care must be taken that the assumption of a zero tracer in the sewer atmosphere remains valid as discussed further down below. A  $\text{SF}_6$  stock solution was obtained by bubbling  $\text{SF}_6$  (99.5%, Carbagas, Switzerland) through

2 L of water for 5 min. One litre of stock solution was diluted 20 times and then pumped into the sewer with a peristaltic pump. An additional easily detectable tracer was required to determine the residence time and for sampling synchronisation. Either rhodamine (visual detection and quantification in water samples by measuring absorption at 554 nm) or a sodium chloride solution (conductometrical detection with a LF 96 conductivity meter, WTW, Weilheim i. OB, Germany) was used. Finally, if infiltration of groundwater was expected then the non-volatile tracer sodium bromide was added continuously with the gas tracer to quantify the amount of dilution according to ISO-standards [34].

#### 3.2. Sampling and analysis

At each sampling location, 10–15 duplicate samples of 25 mL were taken in 50 mL glass syringes (Model ‘Ultrafit’, Henke, Germany) leaving no headspace. Air bubble inclusion was avoided because these lead to erroneous analysis results. The syringes were stored under water until analysis, which was done within 24 h after sampling.

The samples were analysed as described by Schlatter [35]. Nitrogen gas (25 mL) was added to the syringes containing the samples which were vigorously shaken in order to establish an equilibrium between gas and liquid. The headspace was injected on a gas chromatograph at 30°C. It was equipped with an 1 mL sample loop, a 1.8 m steel column ( $\varnothing = 1/8''$ ) filled with 5 Å, 80/100 mesh washed and dried zeolite (Alltech, Socolabo SA, Switzerland) and an electron capture detector (model ECD 800, Fisons, Brechbühler AG, Switzerland) at 350°C, 50 V, a pulse width of 1  $\mu\text{s}$  and a current of 1.5 nA. The carrier gas was nitrogen at 30 mL  $\text{min}^{-1}$ . The linear range reached from the detection limit of 0.03 to 180 ng  $\text{L}^{-1}$ . The reproducibility of duplicate measurements was 15% for concentrations  $> 0.07 \text{ ng L}^{-1}$  and became  $< 5\%$  for concentrations  $> 1 \text{ ng L}^{-1}$ .

The bromide was analysed in the samples taken for the  $\text{SF}_6$  measurement using an ionchromatograph (Model Dionex DX300, Dionex Corporation, Sunnyvale, CA, USA). The reproducibility of the measurements was  $\leq 2\%$  in the range from 2 to 50  $\text{mg}_\text{Br} \text{ L}^{-1}$ .

A wastewater sample taken during each experiment was taken to the laboratory to determine the  $\alpha_{\text{Kr}}$ . Sodium hypochlorite was added to inhibit microbiological respiration and the sample was gassed with nitrogen to remove oxygen present. Finally, the reaeration in a 2 L stirred vessel was compared to that of a clean water sample that had had the same pretreatment.

#### 3.3. Experimental sites

Most sewer experiments were conducted in the main sewer between two rural towns near Zürich in

Switzerland (*Rümlang*). This concrete circular sewer has a diameter ( $\varnothing$ ) of 0.9 m and a slope of  $0.91 \text{ mm m}^{-1}$ . The measurement section started at a sewer drop with a height of 0.5 m that ensured complete mixing of added tracers. Manholes did not influence the water flow as they had circular inverts.

A second site was used in the concrete main sewer near *Bauma* which is located in a rural area in Switzerland. This site has a much higher slope of  $8 \text{ mm m}^{-1}$  and a diameter of 0.5 m.

The third and fourth site contained clean water allowing for combined measurement of oxygen and tracer mass transfer. The third site (*Dorfbach*) was an interconnecting pipe with a total length of 500 m between two sections of a creek. Oxygen consumption can be negligible because no substrate for aerobic or anaerobic microbial processes was present. Two dead-zones, at  $x = 1 \text{ m}$  and between  $x = 179$  and  $189 \text{ m}$  (where  $x$  is the longitudinal coordinate), had a volume of  $1.8$  and  $4.1 \text{ m}^3$ , respectively. The residence time in these dead zones ensured that  $\text{Na}_2\text{SO}_3$  and  $\text{CoCl}_2 \cdot 6\text{H}_2\text{O}$  that were added to remove approx. half of the dissolved oxygen, had finished reacting with oxygen. The measurement section ran from  $x = 189$  and  $309 \text{ m}$ . The material up to the measurement section was vitrified clay pipe ( $\varnothing = 0.250 \text{ m}$ ), within this section it was high density polyethylene ( $\varnothing = 0.315 \text{ m}$ ). The slope was  $14.9 \text{ mm m}^{-1}$ .

The fourth site (*Verbindungskanal*) was a newly constructed circular pipe ( $\varnothing = 2.2 \text{ m}$ ,  $L = 245 \text{ m}$  and  $S_0 = 4.5 \text{ mm m}^{-1}$ ). A construction shaft ( $L \times W \times H = 12.9 \times 2.8 \times 5 \text{ m}$ ) was closed off from the channel with a steel plate and flooded to serve as a clean-water reservoir. Sixteen hours prior to the experiment,  $7 \text{ kg Na}_2\text{SO}_3$  and  $\text{CoCl}_2 \cdot 6\text{H}_2\text{O}$  were dissolved in  $100 \text{ L}$  of water, pumped into the flooded shaft and mixed with pumps overnight to reduce the dissolved oxygen concentration to  $3 \text{ g L}^{-1}$ . Two litres of  $\text{SF}_6$ -saturated water were also added. Measuring and sampling sites were installed at  $x = 20$  and  $220 \text{ m}$ . Oxygen was measured with two OXI 340 (WTW, Weilheim i. OB, Germany) dissolved oxygen meters per site. These have an accuracy of 1% at a flow velocity of more than  $0.1 \text{ m s}^{-1}$ . At the start of the experiment, a valve in the steel plate was opened and water entered the channel at a controlled rate. The flow was determined through the drop in water level in the shaft, measured with a pressure transducer, and with two flows meters within the channel. The sampling was synchronised with the help of a salt pulse. Sampling started 5 min after this pulse had passed at a site and continued for 10 min.

None of these sewer reaches had side connections. In the sewers with a diameter  $\geq 0.5 \text{ m}$ , the water level and discharge were measured with Sigma 950 area velocity flow meters (American Sigma, Medina, NY, USA). Their accuracy of  $< 5\%$  was confirmed with the dilution

method for open channel flow [34] using bromide as a tracer. The dilution method [34] was used in the *Dorfbach*.

## 4. Results and discussion

### 4.1. Tracer concentration in the sewer atmosphere

The  $\text{SF}_6$  concentration in the sewer atmosphere might increase to a level where the assumption of a zero concentration is no longer valid. Consequently, the measured mass transfer will be too small. This is especially true for a step addition.

However, the maximum gas tracer amount added during an experiment was  $\approx 20 \text{ mg SF}_6$ . If this would collect in the sewer atmosphere, which is approx. 90% of the total volume of  $1300 \text{ m}^3$  for the *Rümlang* main sewer, this would have resulted in a concentration of  $17 \mu\text{g m}_{\text{gas}}^{-3}$ . The Henry coefficient for  $\text{SF}_6$  is 132 (dimensionless) at  $15^\circ\text{C}$  [14]. Hence, the gaseous  $\text{SF}_6$  is in equilibrium with  $0.1 \mu\text{g m}_{\text{liquid}}^{-3}$ , which is negligible compared with the applied tracer concentration of  $50\text{--}100 \mu\text{g m}_{\text{liquid}}^{-3}$ . Furthermore, the sewers are normally ventilated to the free atmosphere by applying either slotted manhole covers or ventilation shafts. Sewer aeration is promoted by the drag force that the water exerts on the gas phase [36]. As a result, the assumption of a zero tracer concentration in the gas phase is valid.

### 4.2. Absorption by the sewer biofilm?

Wastewater carries oils and fats that are constantly precipitated on the sewer wall. The high octane–water partition coefficient of  $\text{SF}_6$  (though  $< 1000$  as estimated by Cirpka et al. [14]) bears the danger that the biofilm absorbs a significant portion of the tracer as the tracer flows through the sewer. This would lead to overestimation of the reaeration. Laboratory and real-sewer control experiments were therefore conducted.

In three parallel batch experiments under controlled conditions in the laboratory, the response of the gas exchange rate on the addition of plain water, wastewater and a high concentration of sewer biofilm was compared. The measurable tracer concentration appeared unaffected by the addition of the sewer biofilm.

In the full-scale sewer experiment, a mixture of  $\text{SF}_6$  and bromide ( $\text{Br}^-$ ) was added in the *Rümlang* sewer for  $1 \frac{1}{2} \text{ h}$ . Samples were taken at two different distances from the addition point, as indicated by the distance  $x$  in Fig 1. If there would have been a significant enriched of  $\text{SF}_6$  in fatty deposits, then the dynamic behaviour of the  $\text{SF}_6$ -concentration would be different from that of the salt-tracer bromide that is not preferentially absorbed by fatty deposits. Fig. 1 shows that the  $\text{SF}_6$  and  $\text{Br}^-$

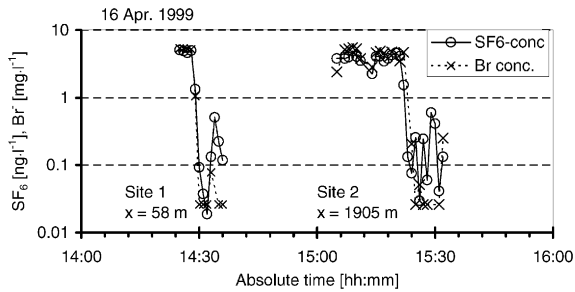


Fig. 1. A comparison between the logarithm of the sulphur hexafluoride (SF<sub>6</sub>) and the bromide (Br<sup>-</sup>)-concentrations (see text for explanation).

concentrations rose and fell in an equal manner showing that no significant absorption and desorption occurred.

It can be concluded that the biofilm in the sewer system will not interfere with the SF<sub>6</sub> gas exchange measurements.

#### 4.3. Selection of boundary layer model

The *Dorfbach* and *Verbindungskanal* experiments were dynamically modelled with the modelling tool AQUASIM2.0A [37]. The hydrodynamic sewer model is based on the Saint Venant equation and the Darcy Weisbach flow equation [38]. This was expanded with gas–liquid mass transfer equation (22) described in this work.

As an example, Fig. 2 shows the concentrations of O<sub>2</sub> and SF<sub>6</sub> during the *Verbindungskanal* experiment. Other data can be found in Tables 2 and 3. The experiments in the *Dorfbach*-channel were processed in the same manner. The results of these experiments showed that the surface renewal theory ( $n = 0.5$ ) prevailed over the film theory ( $n = 1$ ) and should be used in e.g. (7).

### 5. SF<sub>6</sub> sewer measurements

Fig. 3 shows an example of the SF<sub>6</sub> and rhodamine concentration during the gas exchange experiments under steady conditions. The gas tracer SF<sub>6</sub> was added as a step for 45 min. The rhodamine was added as a pulse at 250 m before the first sampling site to synchronise the sampling in the field. A 50 cm high sewer drop 50 m before the first sampling site ensured complete cross sectional mixing.

The sewer dispersion was small (0.1 m<sup>2</sup> s<sup>-1</sup>), which explains the modest widening and lowering of the peak from site 1 to 3. The measured SF<sub>6</sub>-concentrations over a period of 20 min for each site are shown in Fig. 3. The median of concentration is shown with the standard deviations. The time is the average over the sampling time. The inflowing tracer mass was constant. The slight

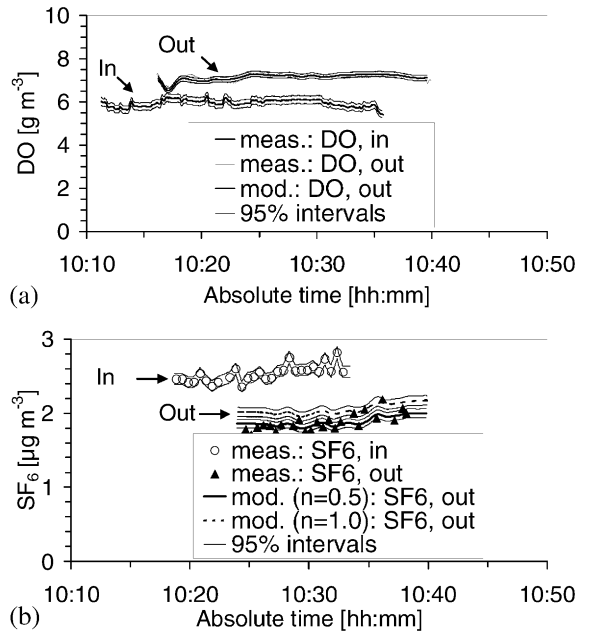


Fig. 2. The measured (meas.) and modelled (mod.) concentrations of dissolved oxygen (DO) (a) and SF<sub>6</sub> (b) during the *Verbindungskanal* experiment. Model results with  $n = 0.5$  and 1 are shown.

increase in the SF<sub>6</sub>-concentration at both sites was caused by a small decrease in flow during the sampling period. The  $k_1a$  is determined by fitting the following equation through the medians:

$$c_{SF_6} = c_0 e^{(k_1 a t)}, \tag{23}$$

where  $c_0$  is the concentration and  $t$  the time. This method proved more reliable than the commonly used method of Brown and Bailod [18] that proved sensitive to variations in the tracer concentration caused by short-lived changes in the discharge, especially if these occurred at the beginning or end of the sampling period.

The conditions during the experiments and the numerical values of the gas transfer rates are listed in the Tables 2 and 3.  $\beta$  has been taken as 0.97.

#### 5.1. Estimation coefficient for sewer reaeration equation

##### 5.1.1. Comparison with previously measured data

The most extensive sewer studies are those of Parkhurst and Pomeroy [20] based on the indirect method and Jensen and Hvitved-Jacobsen [28] who used the direct tracer method with a Dirac pulse addition of the tracers. These data are shown in Fig. 4 together with laboratory data of Moog and Jirka [13] and those acquired during the present study. The data of Balmér and Tagizadeh-Nasser [12] proved to be inconsistent with the other data, probably due to unaccounted

Table 2

An overview of the hydrodynamic conditions and  $k_1 a_{SF_6}$ .  $\tau_r$  = tracer residence time

Location and date	$Q$ (L s <sup>-1</sup> )	$L_{\text{sect.}}$ (m)	$\tau_r$ (hh:mm:ss)	$\bar{u}$ (m s <sup>-1</sup> )	$T$ (°C)	$\alpha_{k_1 a}$ (dimensionless)	$k_1 a_{SF_6}$ (d <sup>-1</sup> )	$k_1 a_{O_2}^{T=20^\circ C}$ (d <sup>-1</sup> )
<i>Rümlang</i>								
12 September 97	28.1	1964	1:30:56	0.36	20.1	0.68	5.4	14.3
14 October 97	24.6	1906	1:12:32	0.44	17.4	0.64	8.3	20.0
17 October 97	14.5	1847	1:31:21	0.34	16.5	0.56	7.1	18.0
16 December 97	13.6	1906	1:25:15	0.37	14.0	0.53	8.0	18.7
07 January 98	49	1847	0:59:45	0.53	13.1	0.76	3.6	8.6
13 August 98	18.9	1788	1:13:46	0.40	20.1	0.69	6.8	17.3
21 August 98	7.7	1788	1:58:15	0.25	19.4	0.57	9.5	18.7
16 April 99	79	1906	0:55:00	0.58	14.0	0.70	4.4	8.1
<i>Bauma</i>								
23 August 98	11.7	1246	0:25:30	0.81	17.2	0.61	45	102
<i>Dorfbach</i>								
17 August 99	1.84	118	0:04:08	0.76	14.3	0.69	74	145
20 August 99	3.96	118	0:02:52	0.90	14.1	0.71	58	92.5
25 August 99	11.6	118	0:01:32	1.28	15.2	0.83	145	238
<i>Verbindungskanal</i>								
2 July 99	12.3	200	0:06:45	0.58	14.2	0.91	33.5	87.9

Table 3

Overview of the calculated hydrodynamic conditions and the resulting *calculated*  $k_1 a_{O_2}$  for the experiments in Rümlang and Bauma (23 August 98)

Eq. No. → Date	$A_{cr}$ (A.1) (m <sup>2</sup> )	$\alpha$ (A.1) (-)	$w$ (A.3) (m)	$P_w$ (A.5) (m)	$d_m$ (A.2) (m)	$R_h$ (A.4) (m)	$u^*$ (20) (m s <sup>-1</sup> )	$Fr$ (21) (-)	$k_1 a_{O_2, calc.}$ (22) (d <sup>-1</sup> )
<i>Rümlang</i>									
12 September 97	0.078	0.88	0.69	0.79	0.113	0.099	0.030	0.34	12.9
14 October 97	0.056	0.78	0.63	0.70	0.080	0.080	0.027	0.49	16.7
17 October 97	0.043	0.71	0.58	0.64	0.068	0.068	0.025	0.41	19.2
16 December 97	0.036	0.67	0.56	0.60	0.061	0.061	0.023	0.48	21.6
7 January 98	0.092	0.93	0.72	0.84	0.110	0.109	0.031	0.51	16.4
13 August 98	0.047	0.73	0.60	0.65	0.071	0.071	0.025	0.48	19.0
21 August 98	0.031	0.63	0.53	0.56	0.054	0.054	0.022	0.35	23.5
16 April 99	0.125	1.04	0.78	0.94	0.16	0.132	0.034	0.46	9.4
<i>Bauma</i>									
23 August 98	0.014	0.49	0.34	0.62	0.023	0.016	0.032	1.69	93.6
<i>Dorfbach</i>									
17 August 99	0.0039	0.63	0.19	0.20	0.021	0.019	0.053	1.11	213
20 August 99	0.0058	0.73	0.21	0.23	0.027	0.025	0.061	1.75	199
25 August 99	0.0090	0.86	0.24	0.27	0.038	0.033	0.070	4.42	256
<i>Verbindungskanal</i>									
2 July 99	0.315	0.32	0.68	0.69	0.036	0.036	0.040	0.68	73.0

oxygen entrainment in their laboratory set-up. This data has therefore not been used.

Moog and Jirka [7] obtained their data under the best controlled conditions and accurate analysis. Still, the scatter in the field data of Parkhurst and Pomeroy [20] is not very much higher, indicating that the Parkhurst and Pomeroy data set is of high quality, although the authors mention several experimental difficulties (small

oxygen concentration differences, limitations of the available measurement techniques, water depth measurement). Furthermore, residual oxygen sinks might have influenced their results.

The data of Jensen [10] have a considerable amount of scatter. For example, two experiments with a discharge difference of only 25% showed a difference in  $k_1 a$  of a factor three. According to the author, this might be



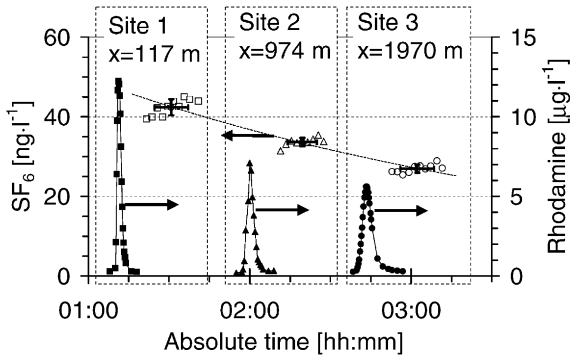


Fig. 3. The measured concentrations of rhodamine as the sampling synchronisation tracer and the SF<sub>6</sub>-concentration shown as a function of the absolute time.  $x$  is the longitudinal distance of the first sampling site from the tracer addition point, etc. (see text for further explanation).

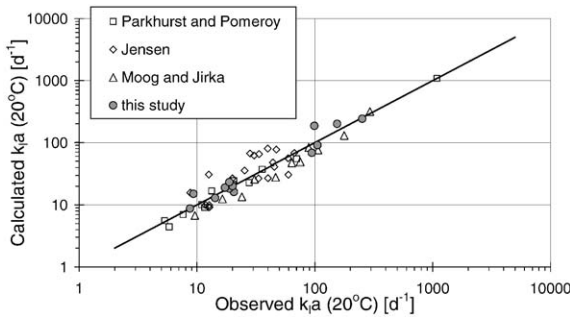


Fig. 4. The observed  $k_1a_{O_2}$  against the value calculated with (25). The line is the diagonal.

caused by the influence of the wall roughness, since uncleaned gravity sewers were used. However, some of the experimental procedures are questionable:

- the flow rate was measured either before or after an experiment, calculated from the water depth. Neither of these methods will give accurate results in small sewer due to the strong discharge fluctuations and the variable relationship between discharge and water depth.
- The tracer was added as pulse and an extra tracer was required to quantify the dispersion.
- The tracer was pumped from the manhole to the surface. However, Schlatter [35] mentions a loss of up to 6% of a gas tracer (sulphur hexafluoride) through the tube wall, depending on the type of material used.

The data of Jensen who originally used the film theory to convert the krypton values to oxygen, are more similar to the data of the other studies when his results are

converted with the surface renewal theory which was found experimentally during this work to be the appropriate boundary layer model.

### 5.2. Parameter estimation

Moog and Jirka [7] showed that the criterium for parameter estimation should not be the minimisation of the residual sum of squares ( $SS_{res}$ ), but of the mean multiplicative error (MME):

$$MME = \exp \left[ \frac{\sum_{i=1}^N |\ln[(k_1a)_p / (k_1a)_m]_i|}{N} \right], \quad (24)$$

where the indices  $p$  and  $m$  are predicted and measured, respectively. The MME gives a more even contribution of errors at small and high reaeration rates.

The coefficients of (22) were fitted with the data shown in Fig. 4. The original data of Parkhurst and Pomeroy [20] were averaged on a per site basis because the original data was presented as several subsequent runs per site under constant conditions. The result of the parameter estimation is

$$k_1a_{O_2}^{T=20^\circ C} = 1500 \frac{u^*(Re^*)^{-1/4} Sc^{-1/2} (1 + 0.5Fr)}{d_m}. \quad (25)$$

The coefficient  $m = -\frac{1}{4}$  gave the optimal results, indicating that the small eddy model was prevailing in conditions commonly found in sewers systems. A MME of 1.22 was obtained. The classic Parkhurst and Pomeroy [20] equation (13), had a MME of 1.27 that was reduced to 1.24 when the parameter were re-estimated. Other equations, like (12) of Thackston and Krenkel [4], led to MMEs > 1.5, even when their constants were redetermined.

### 5.3. Inclusion of Froude number

Fig. 5 shows that in the sewer system  $Fr$  affects  $k_1a$ . When the term with  $Fr$  is omitted and the parameters are redetermined ( $a_{K_1a} = 1900$ ), a MME of 1.4 results. Clearly, the inclusion of  $Fr$  results in an extra fitting parameter, but the effect on MME is practically exclusively observed in the supercritical range, indicating that the  $Fr$ -term has a physical meaning. This can be confirmed with a plot of the residual against  $Fr$  for the case with and without inclusion of the  $Fr$  in (25) (see Fig. 6).

The data of Jensen has an above-average scatter in Fig. 5. This is probably related with the way the discharge was determined (see remarks above). Discarding this data did decrease the MME to 1.19 but had little effect on the parameters.

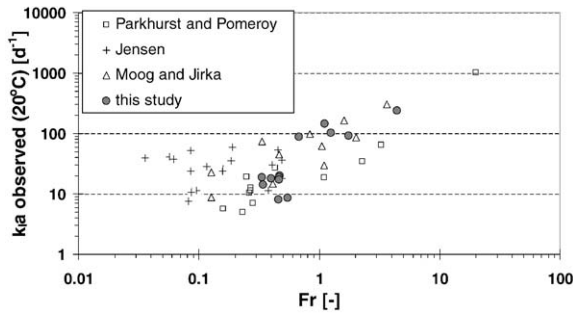


Fig. 5. The Froude number against the observed  $k_{l,a_{20,O_2}}$ .

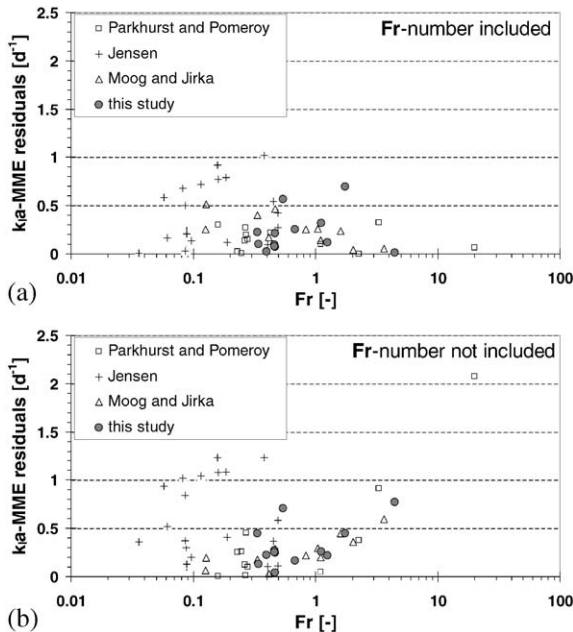


Fig. 6. The  $Fr$ -number against the MME-residuals when (a)  $Fr$  is included in (25) and (b)  $Fr$  not included by setting  $b_{k_1a} = 0$ .

**6. Gas–liquid mass transfer as a function of hydrodynamic and structural variables**

The question might arise what the actual effect of the hydrodynamics and slope of the sewer on the gas–liquid mass transfer is. Therefore, the reaeration has been calculated for an imaginary sewers with (25). The results in Fig. 7 show that the slope is by far more important than the discharge for the  $k_1a$ . The potential reaeration as depicted in Fig. 8 is calculated as

$$\delta c_{O_2} = k_1 a_8 \frac{L_{\text{section}}}{\bar{u}} \tag{26}$$

For a certain channel slope, the amount of oxygen that is transferred into the water is not very strongly affected by the discharge. In other words, Fig. 8

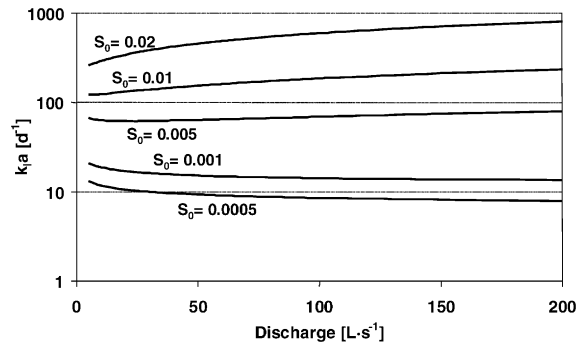


Fig. 7. The  $k_1a$  as a function of discharge and slope ( $S_0$ , in  $m \cdot m^{-1}$ ) in an imaginary circular channel with a diameter of 1 m.

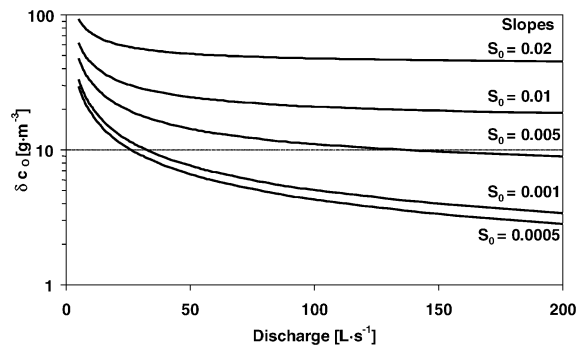


Fig. 8. The calculated reaeration of deoxygenated water over a section of 100 m in the imaginary circular channel as a function of discharge and slope ( $S_0$ , in  $m \cdot m^{-1}$ ) (see also Fig. 7).

indicates that oxygenation of water is not very different whether a sewer with a certain slope carries 50 or 100  $L \cdot s^{-1}$ , values that would be typical for a sewer of this size. However, when the flow is much less, which can occur during the night, the amount of oxygen that can be transferred per litre of water strongly increases. This effect can lead to strong diurnal variations in the oxygen balance in the sewer system [1].

**7. Conclusions**

- A direct (tracer) method to quantify reaeration was adapted and tested for the sewer system. This method where the inert gas tracer  $SF_6$  was added as a step, gave very good results. Several measurements under different flow rates were conducted in four different sewers and pipes.
- In real sewers, the surface hydraulic roughness of the sewer biofilm is important, in addition to larger structures such as uneven links between pipe sections and sediments. Therefore, it remains desirable to

experimentally quantify the gas exchange whenever possible.

- The reaeration model developed for smooth channels (18) by Moog and Jirka [13] is well suited, though a term including the Froude number must be included in order to prevent underestimation of the sewer reaeration at supercritical flow.
- The results of this study and those of former studies by other researchers were used to determine the coefficients of a sewer reaeration equation (25) that is valid for sub- and supercritical flow through smooth and circular open channels, e.g. concrete lined streams and sewers.
- The presence of surfactants is one of the major problems in the quantification of gas–liquid mass transfer. Surfactants can very well be responsible for the observed deviations in the data but their influence will depend on their type and concentration, both of which will change in a diurnal pattern and will be different from one sewer to the next. It is therefore advisable to determine  $\alpha_{k_{1a}}$  when possible. In wastewater  $\alpha_{k_{1a}}$  will be in the range of 0.5–0.9.

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## Appendix A

The angle  $\alpha$  is the angle at the centre of a circular channel between a line perpendicular to the pipe insert and a line towards the point where the water surface touches the pipe wall. It is calculated with an iterative procedure from

$$A_{cr} = 1/4D^2(\alpha - \sin(\alpha)\cos(\alpha)) = Q/\bar{u}, \quad (\text{A.1})$$

where  $D$  is the pipe diameter and  $Q$  the discharge.  $\bar{u}$  can be calculated readily from the residence time of a tracer  $\tau_r$  and the channel length  $L$ . The mean water depth,  $d_m$ , is calculated as

$$d_m = A_{cr}/w, \quad (\text{A.2})$$

where  $w$  is calculated as

$$w = D \sin(\alpha), \quad (\text{A.3})$$

$R_h$  is calculated as

$$R_h = D/4 = A/P_w \quad (\text{A.4})$$

with  $P_w$  as the wetted perimeter

$$P_w = D\alpha. \quad (\text{A.5})$$

The following compound data have been used:

Water dynamic viscosity (0–50°C) [39]

$$\eta_{\text{H}_2\text{O}} = 10^{\frac{1301}{998.333+8.1855(T-20)+0.00585(T-20)^2} - 3.30233}$$

Water density [39]

$$\rho_{\text{H}_2\text{O}} = -1.62 \times 10^{-5}(3.98 - T)^3 - 0.005799(3.98 - T)^2 + 0.024738(3.98 - T) + 1000.$$

Oxygen diffusion [27]

$$\mathcal{D}_{\text{O}_2} = 4.864 \times 10^{-13} T^2 + 2.880 \times 10^{-11} T + 1.268 \times 10^{-9}$$

sulphur hexafluoride diffusion [40]

$$\mathcal{D}_{\text{SF}_6} = 2.9 \times 10^{-6} \exp(1.93 \times 10^{-4}/(RT_K)),$$

where  $R$  is the ideal gas constant (8.31441) and  $T_K$  the absolute temperature.

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