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Empirical modeling of air-to-water oxygen transfer in gravity sewers

Niels Aagaard Jensen

ABSTRACT: To evaluate existing empirical models that have been used for prediction of air-to-water oxygen transfer in gravity sewers, field and laboratory investigations were conducted. A series of 16 oxygen-transfer coefficients were obtained in noncleaned operating sewers using a gaseous tracer method. In addition, experiments on the effect of contaminants on air-to-water gas transfer, expressed through the alpha factor, were conducted in a laboratory vessel. Based on the obtained oxygen-transfer coefficients, five empirical models, all previously used within the sphere of sewer problems, were evaluated. Furthermore, previously reported transfer coefficients for gravity pipes were included in the evaluation. The model of Parkhurst and Pomeroy was found to be the best for simulation of oxygen transfer. Estimates from the worst alternative tend to overestimate the oxygen-transfer coefficient by approximately a factor of 25 on the average. Except for the case when foam was present, the investigations on the alpha factor showed no significant deviation from unity. Finally, the coefficients in the equation of Parkhurst and Pomeroy were evaluated by a nonlinear regression on the available observations from operating sewers. A new set of coefficients based on a statistical evaluation were obtained. *Water Environ. Res.*, **67**, 979 (1995).

KEYWORDS: aeration, alpha factor, dissolved oxygen, gas transfer, gravity sewers, wastewater, tracers.

Gas transfer at the air-to-water interface plays a very significant role for theoretical as well as practical studies of processes in sewers. For example, in the case of in-sewer biodegradation of organic matter the transfer of oxygen from the sewer atmosphere to the flowing wastewater determines if aerobic conditions in the wastewater can be obtained. At low oxygen-transfer rates combined with high-respiration rates, anaerobic conditions may prevail in the sewer, resulting in significant hydrogen sulfide concentrations in the flowing wastewater. Hydrogen sulfide is known to cause several problems, that is, odor, toxicity and corrosion (U.S. EPA, 1985). If hydrogen sulfide is produced, gas transfer will furthermore determine its emission rate. The emission of other toxic air contaminants and odorous substances from public and industrial sewer systems are governed primarily by the air-to-water interface transfer. Thus, recently Corsi *et al.* (1992) published a modeling approach for volatile organic compound (VOC) emissions from sewers. This is probably the first in a row of models describing the fate of specific contaminants in sewer systems.

In general, the dissolved oxygen (DO) level in sewer systems is a key parameter for almost all in-sewer processes. In addition to the hydrogen sulfide problem, oxygen levels within both sewer biofilm, sediment accumulated at the bottom of the sewer, and flowing wastewater influence transformation rates of specific groups of organic matter (Nielsen *et al.*, 1992). Realizing that residence times in sewer systems often are similar to treatment plant residence times, there are reasons to believe that these transformation processes may have a significant effect on the

composition of the wastewater arriving at the treatment plant. Because processes for advanced wastewater treatment, that is, biological nitrogen and phosphorus removal, depend on the quality and quantity of organic matter, the importance of the composition of inflowing wastewater increases. In the future, the ability of modeling a time distribution of quantity and quality of inflowing wastewater will probably be necessary for making the most of advanced treatment processes. No such deterministic computational model will succeed without a satisfactory description of the air-to-water gas transfer.

This work presents a step in the search for a reliable and generally applicable description of air-to-water gas transfer in sewer systems. The investigation was delimited to oxygen transfer in gravity sewer reaches without constructions creating extreme turbulence, such as drop structures. Oxygen-transfer coefficients were measured in operating gravity sewers using the gaseous tracer method of Jensen and Hvitved-Jacobsen (1991). In addition, supplementary data such as chemical oxygen demand (COD), DO, temperature, and flow were determined. Additional laboratory experiments were conducted to determine the effect of water contaminants on gas transfer. The observations obtained were used for evaluation of empirical models that have been proposed for sewer conditions.

Theoretical Background

The overall gas transfer velocity. The flux of a gas across the air-to-water interface is commonly expressed as

$$F = K_L(C_s - C) \quad (1)$$

Where

F = gas flux across the air-to-water interface, $\text{g/m}^2 \cdot \text{h}$;

K_L = overall gas-transfer velocity, m/h ;

C_s = gas saturation concentration, g/m^3 ; and

C = mean concentration of the gas in the water phase, g/m^3 .

C_s is related by Henry's law to the partial pressure of the gas in the atmosphere

$$p = H_c \cdot C_s \quad (2)$$

Where

p = partial pressure in the atmosphere, atm; and

H_c = Henry's law constant, $\text{atm} \cdot \text{m}^3/\text{g}$.

In accordance with the two-film theory of Lewis and Whitman (1924), the overall transfer velocity can be expressed by transfer velocities in the water and gas films, respectively

$$1/K_L = 1/k_L + 1/(H_c \cdot k_g) \quad (3)$$

Where

k_L = gas-transfer velocity in the liquid film, m/h; and
 k_g = gas-transfer velocity in the gas film, g/atm · m² · h.

If the ratio $k_L/(H_c \cdot k_g)$ is significantly smaller than unity, the process described by Equation 3 is water side controlled, and thus the overall transfer velocity is approximately equal to the liquid film velocity; if the ratio is significantly larger than unity, the process is gas side controlled; and if the ratio is near unity, both sides are important. Estimates for k_L and k_g are obtained from gases that are clearly controlled by one side only. If such estimates are used and compared with known Henry's constants, many gases important for environmental engineering phenomena (for example, O₂, N₂, CO₂, and CH₄) are water side controlled. However, some compounds [for example, halogenated hydrocarbons and polychlorinated biphenyl (PCB)] may be in the intermediate range. For these substances the aerodynamic conditions above the surface are equally as important as the hydrodynamic conditions in the water phase. Mass transfer of gases of high solubility and/or high reactivity in the water are controlled by the gas phase (for example, SO₂ and NH₃). This paper focuses on water phase-controlled situations.

The specification of the transfer velocity K_L (equal to k_L for total water side control) is a key element in almost any water-quality model. Therefore, as shown by Bennett and Rathburn (1972), among others, numerous predictive formulations for gas transfer have been presented in the literature. These formulations can generally be split into two classes: conceptual (theoretical) models in which a schematic mechanism for the near-surface interactions of molecular diffusion and the turbulent flow near the surface is taken into account; and empirical models that represent a best fit between observed gas-transfer rates and some global variable, such as channel slope, mean water velocity, or mean water depth. The conceptual models are important as tools to gain understanding of the basic mechanism in the gas-transfer process. However, the difficulty in these models is how to relate the assumed turbulent flow variables near the surface (for example, surface renewal rates) to the overall flow parameters. Therefore, the empirical models are important to cope with engineering problems. Unfortunately, empirical models, in general, lack physical validity, and they cause problems when applied outside their range of applicability.

Conceptual models. The far most generally used of the conceptual models is the previously mentioned two-film theory of Lewis and Whitman (1924). This model assumes uniformly mixed liquid and air phases separated by thin films in which gas transfer takes place by molecular diffusion. Thus, on the premise of water phase control, it can be found from the two-film theory that

$$K_L = D/L \quad (4)$$

Where

D = molecular gas-diffusion coefficient in water, m²/h; and
 L = liquid film thickness, m.

Another classic description of the gas-transfer process is the surface renewal theory. In reality, this theory is a group of models based on penetration theory (Higbie, 1935). The basic idea in Higbie's penetration theory is that turbulent flow in the water body produces parcels of fluid which are in direct contact with the air phase for a short, constant period of time before being

mixed with the bulk water. During the exposure time, molecular diffusion results in penetration of gas into the parcel to a depth which is small compared with the parcel.

Danckwerts (1951) replaced the assumption of constant exposure time used by Higbie with a surface-age distribution function, ϕ , based on a fractional renewal rate, r .

$$\phi = r \cdot \exp(-r \cdot t) \quad (5)$$

Where

ϕ = surface-age distribution function, —;
 r = fractional rate of surface renewal, s⁻¹; and
 t = time, s.

Based on these considerations Danckwerts found the gas-transfer velocity to be

$$K_L = (D \cdot r)^{0.5} \quad (6)$$

In both the Lewis and Whitman model and the Danckwerts model, K_L is proportional to D^n . However, n has the value of 1 for two-film theory contrary to 0.5 for surface renewal theory of Danckwerts. In the film-penetration theory of Dobbins (1956), a combination of the previous models was obtained, resulting in a variable value of the exponent n . Combining the boundary conditions of a liquid film with thickness, L , with the Danckwerts' surface-age distribution function, Dobbins (1956) found that

$$K_L = (D \cdot r)^{0.5} \coth(r \cdot L^2/D)^{0.5} \quad (7)$$

In case of r approaching zero, Equation 7 equals Equation 4. Likewise Equation 7 equals Equation 6 when r approaches infinity.

More surface renewal models and other models with different analytical approaches have been presented in the literature. A comprehensive review can be found in Bennett and Rathburn (1972) and Theofanous (1984).

Empirical models. For practical purposes an overall gas-transfer coefficient, $K_L a$, defined as

$$K_L a = K_L \cdot a = K_L \cdot A/V = K_L \cdot d_m^{-1} \quad (8)$$

Where

a = ratio of water-to-air surface area, A , to volume of water, V, m⁻¹; and
 d_m = mean depth of the water phase, m⁻¹,

is more relevant than the gas-transfer velocity, K_L . Thus, for environmental engineering purposes the rate of gas transfer is generally described by the following equation, valid for a batch reactor with no other gas consuming or producing processes involved.

$$dC/dt = \alpha \cdot K_L a (\beta \cdot C_s - C) \quad (9)$$

Where

C = gas concentration in the liquid phase at the time, t , g/m³;
 $K_L a$ = overall gas-transfer coefficient for clean water, h⁻¹;
 α = ratio of $K_L a$ in polluted water to that in clean water, —;
 C_s = saturation concentration of the gas in clean water, g/m³;

Table 1—Empirical models proposed for estimation of gas-water oxygen transfer in gravity sewer systems.

Reference	Equation	No.
Krenkel and Orlob (1962)	$K_L a(20) = 7.235 (u \cdot s)^{0.406} d_m^{-0.66}$	(10)
Owens <i>et al.</i> (1964)	$K_L a(20) = 0.222 u^{0.67} d_m^{-1.85}$	(11)
Parkhurst and Pomeroy (1972)	$K_L a(20) = 0.96 (1 + 0.17 F^2) (s \cdot u)^{3/8} d_m^{-1}$	(12)
Tsivoglou and Neal (1976)	$K_L a(20) = B \cdot u \cdot s$	(13)
Taghizadeh-Nasser (1986)	$K_L a(20) = 0.4 u (d_m/R)^{0.613} d_m^{-1}$	(14)

Where

$K_L a(20)$ = gas(oxygen)-transfer coefficient at 20°C, h⁻¹;

u = mean velocity of flow, m/s;

s = slope, m/m;

d_m = hydraulic mean depth, defined as the water cross section area divided by the width of the water surface, m;

R = hydraulic radius, m;

F = Froude number, $F = u/(g \cdot d_m)^{0.5}$;

g = gravitational acceleration, m²/s²; and

B = a coefficient, given as a function of water quality and intensity of mixing, dimensionless (here 720).

β = ratio of solubility in polluted water to that in clean water, —; and

t = time (h).

The five empirical models for gas-water oxygen transfer presented in Table 1 have been used within the sphere of sewer system problems. For the purpose of consistency, all models presented have been rewritten to express $K_L a$ (base e) in hour⁻¹ at 20°C. The temperature correction is expressed by the use of

$$K_L a(20) = K_L a(T) \cdot \theta^{(20-T)} \quad (15)$$

Where

$K_L a(T)$ = $K_L a$ at temperature T , h⁻¹;

T = water temperature, °C; and

θ = exponential temperature coefficient, —.

The value of 1.024 for θ determined by Elmore and West (1961) has been used because it has become a broadly accepted value within environmental engineering. In reality, θ may be a function of turbulence as reported by Jensen (1991).

The models presented by Krenkel and Orlob (1962) and Owens *et al.* (1964), Equations 10 and 11 in Table 1, have been proposed in Thistlethwayte (1972) for use in relation to estimation of hydrogen sulfide problems in sewers. However, these models, as will be discussed later, were derived for conditions different from those generally found in gravity sewer systems.

Until now only the model by Parkhurst and Pomeroy (1972) has been derived from investigations carried out in operating gravity sewers. Parkhurst and Pomeroy presented 74 oxygen-transfer coefficients determined in 12 operating municipal sewers in Los Angeles using a mass-balance method. The investigated sewers (diameter 420 to 1 220 mm, length 420 to 3 200 m, and flow 0.015 to 0.250 m³/s) were cleaned mechanically and treated with both caustic soda and hypochlorite to eliminate the biological activity. The DO concentrations were determined at the two ends of each test reach using oxygen electrodes, and oxygen-transfer coefficients were then determined by the mass-balance principle. On the basis of an evaluation of the conducted sewer observations and previously published data on reaeration rates in streams, Parkhurst and Pomeroy concluded that the oxygen-transfer coefficient for gravity sewers can be represented by

Equation 12. This model was applied by U.S. EPA (1985) among others in relation to hydrogen sulfide problems. It is important to notice that contrary to the other equations, the α coefficient is indirectly included in the general constant, 0.96, in Equation 12.

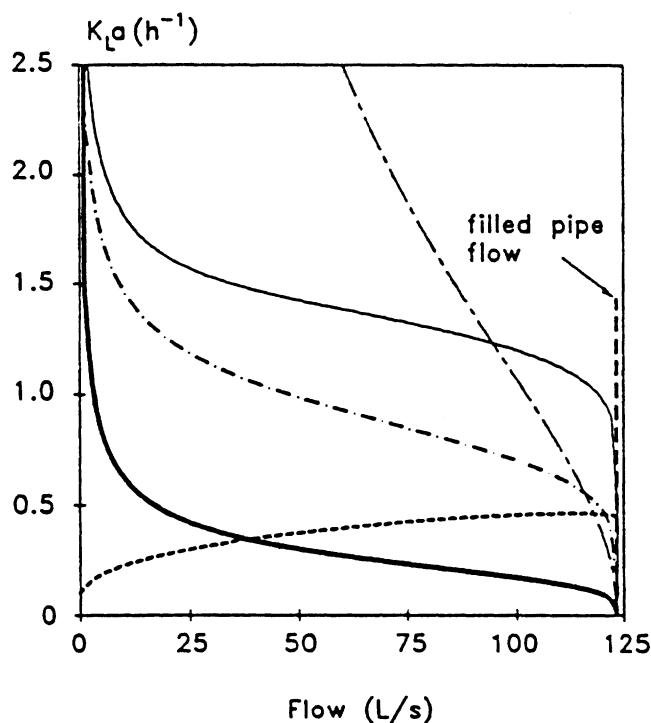
Equation 13 (Tsivoglou and Neal, 1976) is included to illustrate the risk of logical error when applying empirical models on systems different from what they originally have been developed for. This will be discussed later. The empirical model of Tsivoglou and Neal has been used in Meyerhofer *et al.* (1990) for estimation of free surface volatilization of VOCs from conveyance channels in wastewater treatment plants.

In Taghizadeh-Nasser (1986) 50 experiments were conducted in a 225-mm-diameter, 24-m-long laboratory test pipe using tap water. Several relations between hydraulic parameters and the determined transfer coefficients were considered. Equation 14 was presented as an example with a correlation coefficient of 0.97 between observed and predicted values.

If the five empirical models from Table 1 are used on a given sewer pipe, there should be consensus on the forecast. In Figure 1 a typical example of a comparison is shown. The sewer chosen for construction of Figure 1 corresponds to a normal Danish intercepting sewer. Knowing diameter, slope, roughness, and temperature, the missing values of flow velocity and mean hydraulic depth were found as a function of flow by using the relevant hydraulic equations. There is clearly quite an inconsistency between the five models shown in Figure 1. Therefore, it is the objective of this paper to present new data obtained from operating sewers and to use these for evaluation of the empirical models.

Experimental Procedures

Method. The direct method of Tsivoglou *et al.* (1965) using radioactive tracers was selected for both field and laboratory investigations. The method has been applied by Jensen and Hvitved-Jacobsen (1991) for measurements in gravity sewers. Basically, the direct tracer method involves the use of a non-reactive, gaseous tracer allowing the exclusion of any other process than air-to-water transfer. When the medium is wastewater, this is of great importance. Krypton-85 is used as a tracer for the air-to-water gas transfer, and tritium is used as a dispersion



- Krenkel and Orlob (1962)
- Owens et al. (1964)
- Tsivoglou and Neal (1976)
- Taghizadeh-Nasser (1986)
- Parkhurst and Pomeroy (1972)

Figure 1—Estimated oxygen-transfer coefficients as a function of flow in a gravity sewer; diameter $d = 0.5$ m; slope $s = 0.001$; roughness $k = 0.001$ m; and temperature $T = 20^\circ\text{C}$.

tracer. When these substances have been released simultaneously upstream in a sewer stretch, and krypton-85:tritium concentration ratios in the flowing wastewater have been determined at two sampling stations downstream, the krypton-85 transfer coefficient is determined by

$$K_L a_{kr} = -\ln(R_d/R_u)/\Delta t \quad (16)$$

Where

- $K_L a_{kr}$ = krypton-85 transfer coefficient, h^{-1} ;
- R_d = krypton-85:tritium concentration ratio in downstream sampling station, —;
- R_u = krypton-85:tritium concentration ratio in upstream sampling station, —; and
- Δt = time of transport between the sampling stations (h).

The accuracy of the gaseous tracer method used in gravity sewers is discussed in Jensen and Hvitved-Jacobsen (1991). A fundamental assumption for the tracer method is that krypton-85 in the gas phase can be neglected. In some situations krypton accumulation in the headspace may lead to a significant error in the determined gas-transfer coefficient. Therefore, each situation should be evaluated carefully.

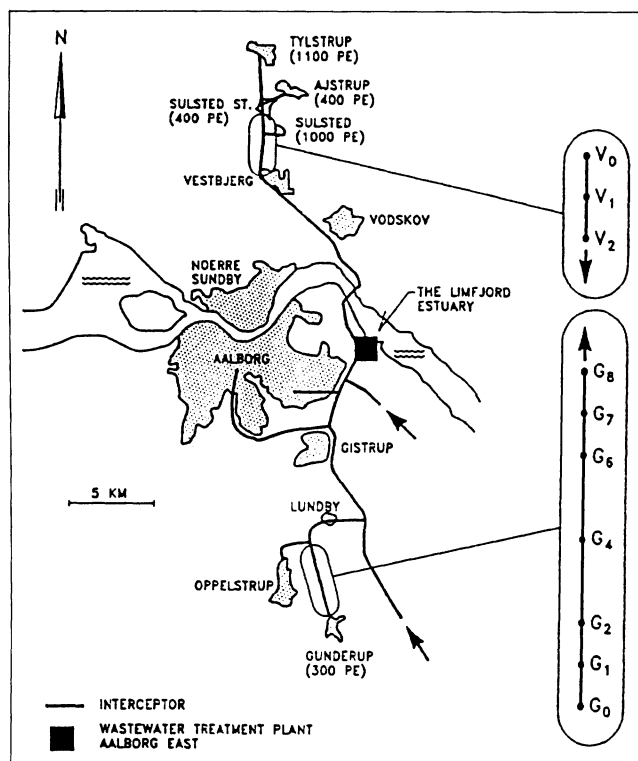


Figure 2—Sites used for field investigations. Data for slope, diameter, and length are found in Table 2.

Knowing $K_L a_{kr}$, corresponding gas-transfer coefficients for other water phase controlled gases can be estimated from the assumption that,

$$K_L a_x / K_L a_{kr} = \text{constant} \quad (17)$$

Where

$K_L a_x$ = gas-transfer coefficient for a given gas, x , h^{-1} .

The constant in Equation 17 can be determined by experiments or by theoretical considerations based on conceptual models (for example, Equations 4, 6, and 7).

Field investigations. Field experiments were carried out at two sites, Figure 2, representing a major and minor Danish gravity sewer pipe, respectively. The first one, the Vestbjerg pipe, is located north of the city of Aalborg, serving approximately 3 000 person equivalents (PE). It was constructed in 1984. The second one, the Gunderup pipe, is located south of Aalborg, serving approximately 300 PE. This pipe was constructed in 1974. Both pipes are located in combined sewer areas receiving mainly wastewater from households. During dry weather approximately only 5% to 10% of the pipe flow capacity is used, which is a typical situation in Danish sewer pipes. In Vestbjerg, two sampling stations, V1 and V2, and a dosing point, V0, 950 m upstream V1 were used. In Gunderup, six sampling stations (G1, G2, G4, G6, G7, and G8) and a dosing point, G0, 230 m upstream G1 were used. The material of the Vestbjerg pipe is concrete, and the material of the Gunderup pipe is plastic (polyvinylchloride). The sampling and handling procedures for the tracer experiments followed the procedures outlined in details in Jensen and Hvitved-Jacobsen (1991).

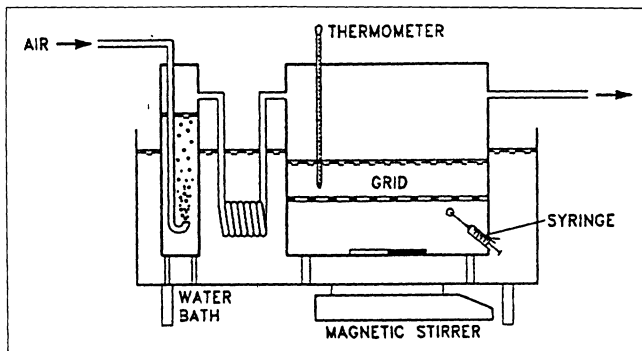


Figure 3—Vessel used for laboratory experiments.

During tracer experiments, supplementary samples were taken at the sampling stations and analyzed for total COD, soluble COD, suspended solids (SS), and conductivity. The DO concentration and pH in the flowing wastewater were directly measured by electrodes. Furthermore, the partial pressure of oxygen in the sewer atmosphere was determined by a gas monitor. This was done as the first procedure when removing a manhole cover to minimize the inevitable ventilation related to the succeeding test procedures.

The flow was not directly measured during tests. However, before and after the Vestbjerg tests, a 90-deg, V-notch, sharp-crested weir was placed just downstream of the investigated pipe section. From the determined flow rates combined with measurements of water depths and velocities during tests the flow was estimated. At the Gunderup site no weir was used, so the flow was estimated from a combination of measured flow depths and determined velocities estimating a roughness parameter for the pipe.

Laboratory investigations. To study the effect of pollutants on gas-transfer coefficients found in the investigated gravity sewers, parallel experiments were conducted in the laboratory. These experiments were performed in a circular, low permeability polyvinylchloride vessel with an inside diameter of 0.265 m and a height of 0.25 m, Figure 3. In all experiments the water phase was mixed with a cylindric magnetic stirrer with length 80 mm and diameter 10 mm. The stirring rate was kept at 300 rpm in all tests. From preliminary experiments this stirring rate was known to give values of $K_L a$ typical for several of the real sewer observations. To avoid a major vortex at the surface, a grid was placed 4 cm under the surface. Samples for analyses were taken with syringes through butyl rubber stoppers placed at the side wall of the reactor. Using a thermostatic water bath the temperature could be held approximately constant ($\pm 0.5^\circ\text{C}$).

At the beginning of an experiment, the reactor was filled with 6 L of the investigated media (collected wastewater or tap water mixed with surfactants). Then a portion of the radioactive krypton-85 gas (approximately 1 μCi) dissolved in tap water (approximately 1 mL) was added to the water phase through the rubber stoppers. After 2 min of mixing, 1 mL of samples was taken with 5- to 10-min intervals using syringes. Immediately after sampling, the volume of sample was transferred to and mixed with scintillation liquid in a scintillation vial. The krypton-85 activity in the samples was determined by multichannel liquid scintillation technique in accordance with Jensen and Hvitved-Jacobsen (1991). No dispersion tracer was added as the reactor

was uniformly mixed. Based on the measured activity, A_{kr} , corrected to a standard quenching level and plotted versus time, the gas-transfer coefficient was calculated using the log-deficit method, which in the case of krypton-85 results in

$$\ln(A_{kr,t}) = \ln(A_{kr,t=0}) - K_L a_{kr} \cdot t \quad (18)$$

Because of the small volume of sample drawn from the reactor during a test, the reduced depth of water in the reactor was neglected in the calculations.

Experimental Results

Field investigations. Table 2 shows the gas-transfer coefficients obtained in the two investigated sewers. Test Nos. 1 to 4 were carried out at the Vestbjerg pipe during daytime (from 9 a.m. to 4 p.m.) referring to normal dayflow. The unexpected low flow in Test No. 1 was expected to be due to interruptions in the upstream sewer system, but no such interruptions were detected by the public sewage works department. Tests Nos. 5 and 6 were carried out early in the morning (from 5 to 9 a.m.) to cover a low nightflow situation. By this procedure, it was the intention to obtain minor flows referring to $K_L a$ values closer to the ordinate axis in Figure 1. However, because of high infiltration in the upstream system, the daily variations in flow were small, which can be seen from Test No. 5. Consequently, a central pumping station in the upstream system was shut down during Test No. 6, resulting in the desired low-flow condition.

Test nos. 7 to 15 were carried out at the Gunderup pipe under normal dayflow conditions. It was found feasible to run some of the tests (Nos. 8 to 10 and 12 to 15, respectively) as consecutive sampling after one release of tracers.

Because all the flows from the first 15 tests carried out at the two pipes were found in the order of 5% to 10% of the full flowing capacity, the possibility to make an artificial increase of flow was considered. Because of the pipe dimensions and the necessary volume of water needed, this was found impossible for the Vestbjerg pipe. However, by pumping water from a local village pond it was possible to artificially increase the flow rate at the Gunderup pipe for sufficient time for measurements. Test Nos. 16 to 20 were obtained under such high-flow conditions. In Table 2 values of the corresponding oxygen-transfer coefficients, $K_L a_{ox}$, have been included to compare these with the empirical models previously listed. The ratio $K_L a_{kr}:K_L a_{ox} = 0.83$ determined by Tsivoglou *et al.* (1965) was used for these estimates. According to Neal (1979) this ratio is unaffected by pollutants.

In connection to all tests the possible error due to accumulation of krypton-85 in the sewer atmosphere was evaluated in accordance with Jensen and Hvitved-Jacobsen (1991). In all cases the possible error was estimated to less than 1%.

When estimating flows, it was realized that the roughness of the pipes was significantly different for the individual pipe sections. Furthermore, it was seen that the roughness of the Gunderup pipe was remarkably reduced during the artificial flow situation. This is most likely a result of flushing of coarse deposits. The values for roughness listed in Table 2 were estimated by use of measured depths and water velocities combined with the relevant hydraulic equations.

As previously mentioned, supplementary samples were taken during tracer experiments. This was done to characterize the wastewater involved in the tests. Based on the values obtained

Table 2—Data summary for field investigations.

Test No.	Pipe section	Diameter, mm	Length, m	Slope, %	Flow, 10 ⁻³ , m ³ /s	Roughness, mm	Velocity, m/s	Residence time, min	Temperature, °C	K _L a, kr, h ⁻¹	K _L a, ox, h ⁻¹	K _L a, ox (20), h ⁻¹
1	V1-V2	600	1358	0.09	7.00	4.0	0.233	97	12.9	0.241	0.290	0.344
2	V1-V2	600	1358	0.09	21.00	4.0	0.323	70	12.8	0.352	0.424	0.503
3	V1-V2	600	1358	0.09	20.00	4.0	0.319	71	11	0.348	0.419	0.519
4	V1-V2	600	1358	0.09	20.00	4.0	0.319	71	11	0.351	0.423	0.524
5	V1-V2	600	1358	0.09	16.00	4.0	0.298	76	9.6	0.328	0.395	0.506
6	V1-V2	600	1358	0.09	1.00	4.0	0.143	158	7.1	0.710	0.855	1.162
7	G4-G6	152.8	400	0.3	0.21	30.0	0.101	66	12.4	1.298	1.564	1.873
8	G2-G4	152.8	481	0.3	0.64	5.0	0.198	40.5	14.1	0.836	1.007	1.159
9	G4-G6	152.8	400	0.3	0.57	30.0	0.136	49	13.2	1.211	1.459	1.714
10	G6-G8	152.8	690	0.3–0.49	0.60	—	0.155	74	12	1.639	1.975	2.387
11	G6-G8	152.8	412	0.49	0.77	10.0	0.222	31	11.4	1.854	2.234	2.739
12	G1-G2	152.8	231	0.3	0.49	10.0	0.164	23.5	15.3	0.919	1.107	1.238
13	G2-G4	152.8	481	0.3	0.55	5.0	0.189	42.5	14.2	0.775	0.934	1.071
14	G4-G7	152.8	678	0.3	0.55	—	0.111	102	12.7	1.247	1.502	1.786
15	G7-G8	152.8	412	0.49	0.56	10.0	0.202	34	11.5	1.103	1.329	1.626
16	G2-G6	152.8	881	0.3	5.10	0.1	0.544	27	10	1.112	1.340	1.698
17	G2-G6	152.8	881	0.3	5.10	0.1	0.544	27	9.8	0.554	0.667	0.850
18	G6-G8	152.8	690	0.3–0.49	5.10	—	0.511	22.5	9.7	0.925	1.114	1.423
19	G2-G6	152.8	881	0.3	4.00	0.1	0.489	30	10.1	1.652	1.990	2.517
20	G6-G8	152.8	690	0.3–0.49	4.00	—	0.460	25	9.9	0.349	0.420	0.534

V = Vestbjerg, G = Gunderup.

(120 to 470 mg COD_{total}/L, 25 to 210 mg COD_{soluble}/L, 60 to 160 mg SS/L, and 70 to 85 mmho/m), the wastewater in general can be characterized as diluted compared with average Danish wastewater [530 mg COD_{total}/L, 210 mg COD_{soluble}/L, 300 mg SS/L, and 100 mmho/m (Henze, 1982)]. However, samples collected during Test No. 11 showed a remarkably high level of COD (4 000 mg COD_{total}/L and mg 480 COD_{soluble}/L). In all

tests pH was found in the interval 7.5 to 7.7, which is normal for Danish wastewater. The DO was found from 0.8 to 4.2 mg/L, which shows that aerobic conditions prevailed during all tests. Values of the oxygen partial pressure in the atmosphere were found from 17.3% to 20.6% immediately after the removal of the manhole cover. However, because of natural ventilation the oxygen partial pressure increased rapidly shortly after.

Table 3—Data summary for laboratory experiments.

Test	Wastewater from	Referring to test number	Temperature, °C	K _L a, kr, h ⁻¹	K _L a, kr (20), h ⁻¹	Alpha
a	Aalborg East		15.3	0.451	0.530	1.060
b	Aalborg East		12.5	0.349	0.452	0.903
c	Aalborg East		13.1	0.405	0.514	1.027
d	Aalborg East		12	0.339	0.446	0.893
e	Aalborg East		12	0.35	0.461	0.922
f	Aalborg East		13	0.421	0.536	1.071
g	Aalborg East		9.8	0.331	0.470	0.940
h	Aalborg East		3.3	0.283	0.503	1.005
i	Aalborg East		2	0.273	0.507	1.014
j	Vestbjerg		16.8	0.462	0.516	1.032
k	Vestbjerg		17.5	0.479	0.522	1.044
l	Vestbjerg	6	13.5	0.435	0.544	1.088
m	Gunderup	11	19.5	0.447	0.455	0.910
n	Gunderup	12–15	18.5	0.543	0.572	1.144
o	Gunderup	16	11.7	0.394	0.524	1.048
p	Gunderup	19–20	12.6	0.429	0.553	1.107
Average alpha			Standard deviation			
All data			1.01			
Aalborg East			0.98			
Vestbjerg			1.06			
Gunderup			1.05			

Table 4—Data summary for surfactants experiments.

Product	Remarks	Temperature, °C	$K_L a$, kr, h ⁻¹	$K_L a$, kr (20) h ⁻¹	Alpha
Wash. powd. (#1)	No foam	19.7	0.504	0.509	1.018
Wash. powd. (#2)	No foam	16.8	0.462	0.516	1.032
Wash. powd. (#2)	0–2 mm foam	18.3	0.666	0.706	1.412
Wash. powd. (#2)	No foam	19.5	0.498	0.507	1.013
Shampoo (#1)	>2 mm foam	19.8	1.134	1.142	2.284
Shampoo (#1)	0–2 mm foam	20	0.684	0.684	1.368
Shampoo (#1)	No foam	20.1	0.504	0.502	1.005
Shampoo (#2)	>2 mm foam	24.2	0.99	0.857	1.714
Shampoo (#2)	0–2 mm foam	23.5	0.708	0.628	1.255
Shampoo (#2)	No foam	22.5	0.606	0.556	1.112
Shampoo (#2)	No foam	16.4	0.492	0.557	1.114
Shampoo (#2)	0–2 mm foam	16.5	0.54	0.609	1.218
Shampoo (#2)	0–2 mm foam	16.8	0.822	0.918	1.835
Shampoo (#2)	No foam	16.5	0.486	0.548	1.096
Shampoo (#2)	>2 mm foam	19.4	1.044	1.066	2.132
Shampoo (#2)	0–2 mm foam	19.5	0.75	0.763	1.526
Shampoo (#2)	No foam	19.6	0.546	0.554	1.107
Average alpha			Standard deviation		
No foam		1.06	0.05		
0–2 mm foam		1.44	0.23		
>2 mm foam		2.04	0.30		

Laboratory investigations. Table 3 shows the results of the alpha factor tests. In addition to the seven tests that were conducted on wastewaters collected at the investigated sites, Vestbjerg and Gunderup, a series of nine tests conducted on wastewaters collected at the inlet of the centralized wastewater treatment plant Aalborg East, Figure 2, were included. Both investigated sewer pipes are connected to this 100 000 PE treatment plant. The inlet wastewater to the Aalborg East Wastewater Treatment Plant can be characterized as normal compared with average Danish wastewater (Henze, 1982). Wastewaters were collected on the day of test except from Tests h and i, which were collected on the day before and kept at 2°C overnight. To estimate alpha factors, the conducted wastewater tests were compared with a series of tap water tests previously published in Jensen (1991). These tap water tests were made under corresponding conditions in the same laboratory vessel. Because the tests published in Jensen (1991) showed a temperature coefficient of 1.035 specific for the laboratory vessel in use, this value was selected for temperature correction of the laboratory tests, even though it differs from the generally accepted value of 1.024 previously mentioned and used for the field tests. The tap water tests showed an average $K_L a_{kr}(20)$ value of 0.500 h⁻¹ with a standard deviation of 0.031 h⁻¹. Because no significant differences on measured krypton-transfer coefficients were observed between the three different origins of wastewaters, the tests were regarded as one series. The conducted Tests a–p show an average $K_L a_{kr}(20)$ value of 0.507 h⁻¹ with a standard deviation of 0.039 h⁻¹. If alpha factors are estimated using the determined $K_L a_{kr}(20)$ value for tap water, an average value of 1.01 ± 0.08 is found, Table 3.

Because surfactants are known to affect the alpha factor significantly (WPCF/ASCE, 1988), supplementary laboratory tests were conducted. From a major Danish manufacturer of detergents, information was achieved on the most important sources to surfactants in Danish domestic wastewaters. Washing powder

and shampoo were recognized as the dominating sources. The manufacturer was asked to produce samples of each with a typical composition for products on the market. An average daily use per person was estimated and compared with the typical daily flow per person. Solutions with these expected typical wastewater concentrations (referred to as No. 1) were produced, and tests were made in the laboratory vessel. Furthermore, solutions with concentrations that were increased with a factor of five (No. 2) were also used for tests. It was seen that the $K_L a_{kr}$ values could be highly influenced by layers of foam on the water surface, Table 4. Therefore, the tests were roughly divided into three groups: tests without foam on the surface; tests with 0 to 2 mm foam; and tests with foam layers thicker than 2 mm. The corresponding alpha factors determined were 1.06 ± 0.05 ; 1.44 ± 0.23 ; and 2.04 ± 0.30 , respectively.

Discussion

The oxygen-transfer coefficients. To evaluate the empirical models for their ability to predict the oxygen-transfer coefficient, calculated values of $K_L a$ obtained from these models have been plotted in Figure 4 against the corresponding observed $K_L a$ values from the field investigations. The data from tests on pipe section G6 to G8 were excluded because of computational problems that arose from the combination of two different pipe slopes. Furthermore, Test No. 14 was excluded because estimation of the pipe roughness was found to be impossible.

All evaluations of accuracy of prediction for the individual empirical models were based primarily on the information obtained through estimated values plotted versus observed values. This visual information was supplemented by two statistical measures, knowing that all statistical measures do have limitations. The goodness of fit was evaluated by means of the following standard error expression:

$$SE = (1/n \cdot \sum_{i=1}^n [\ln(K_L a_{esti}) - \ln(K_L a_{obs})]^2)^{0.5} \quad (19)$$

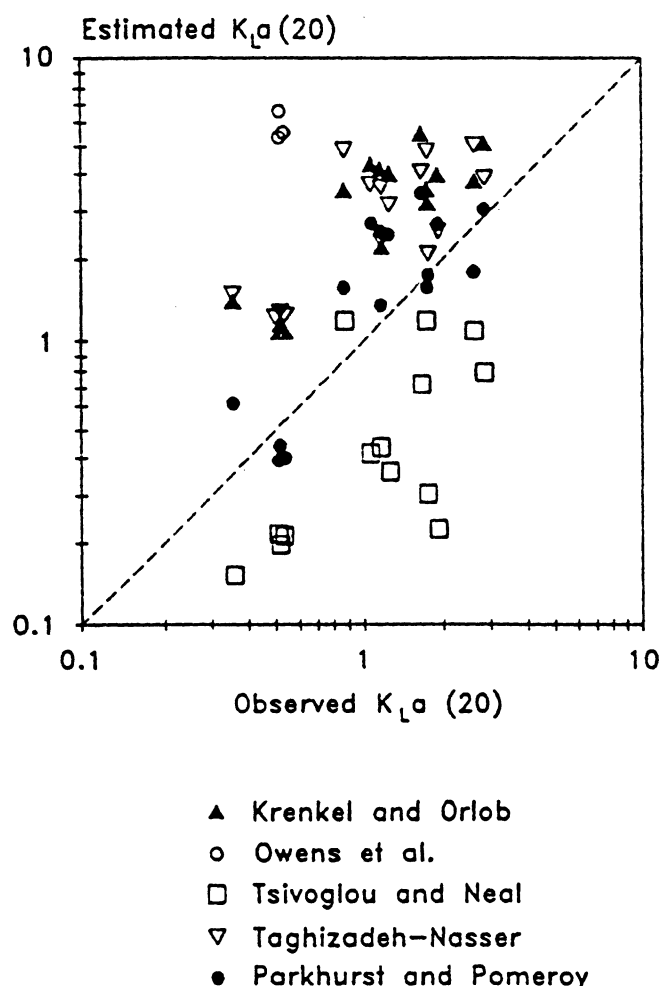


Figure 4—Observed oxygen-transfer coefficients (h^{-1}) obtained in this investigation plotted versus corresponding predicted values.

Where

$K_{La_{esti}}$ = estimated K_{La} value;

$K_{La_{obs}}$ = observed K_{La} value; and

n = number of values in the summation.

Furthermore, a measure of the tendency of under- or overestimation was achieved by the following residual average expression:

$$RA = 1/n \cdot \sum_1^n [\ln(K_{La_{esti}}) - \ln(K_{La_{obs}})] \quad (20)$$

As seen from Figure 4 the model of Owens *et al.* (1964) resulted in estimated values significantly higher than the corresponding observed values when applied on the 16 observations obtained through this investigation. In Table 5 the residual average was found to be 3.26, which can be interpreted as an overestimation by a factor 26 on average. Therefore, most of the data outrange Figure 4. Without doubt, the Owens equation would clearly overestimate the oxygen-transfer coefficient if applied to gravity sewers as proposed by Thistlethwayte (1972). Also, the Krenkel-Orlob equation mentioned by Thistlethwayte results in estimated values beyond the observed values. This is supported by the calculated residual average in Table 5. The points in Figure 4 and the statistical measures in Table 5 representing the model of Taghizadeh-Nasser (1986) are comparable with the ones of Krenkel and Orlob. Also, the Taghizadeh-Nasser equation results in estimated values higher than the observed. The model of Tsivoglou and Neal (1976) gave, on the other hand, estimated values that, in general, were significantly smaller than the observed values. Only one estimated value was found higher than the corresponding observed value. The model of Parkhurst and Pomeroy (1972), on the other hand, gave reasonable estimates of the observed K_{La} values. This is supported by the statistical measures listed in Table 5. However, the residual average of 0.20 indicates that the equation tends to overestimate the oxygen-transfer coefficient by 22% on average.

Table 5—Statistical measures of the empirical equations.

Data source	Krenkel and Orlob (Eq. 10)		Owens <i>et al.</i> (Eq. 11)		Parkhurst-Pomeroy (Eq. 12)		Tsivoglou and Neal (Eq. 13)		Taghizadeh-Nasser (Eq. 14)		Improved equation (Eq. 22)	
	SE	RA	SE	RA	SE	RA	SE	RA	SE	RA	SE	RA
This investigation (16 data)	0.94	0.88	3.33	3.26	0.47	0.20	1.24	-1.07	0.96	0.88	0.43	0.10
Complete series (140 data)	0.89	0.57	2.29	2.17	0.48	-0.14	0.99	0.27	1.17	0.82	—	—
Taghizadeh-Nasser (50 data)	0.19	-0.15	1.85	1.79	0.69	-0.54	0.63	-0.07	0.10	-0.02	—	—
Parkhurst-Pomeroy (74 data)	1.14	1.00	2.27	2.18	0.27	0.06	1.12	0.79	1.55	1.38	0.26	-0.03
Parkhurst-Pomeroy (*) (67 data)	1.19	1.15	2.37	2.33	0.27	0.07	1.08	0.73	1.62	1.57	0.27	-0.03
Operating sewers (90 data)	1.11	0.98	2.49	2.37	0.31	0.08	1.15	0.46	1.46	1.29	0.30	<0.01
Operating sewers (*) (83 data)	1.14	1.10	2.59	2.51	0.32	0.09	1.12	0.38	1.52	1.44	0.31	<0.01

(*)Refers to series in which seven extreme K_{La} values were excluded.

A data set of 16 observed values is probably insufficient for an evaluation of the empirical models. Therefore, the available literature was examined to find published observations of gas-transfer coefficients in gravity pipes. The only useful observations found were the ones of Parkhurst and Pomeroy (1972) and Taghizadeh-Nasser (1986). These data were included for further considerations. Thereby a data set having a total of 140 K_La values all obtained in gravity sewer pipes were available. The complete set of data consists of three different types of observations:

- observation obtained in noncleaned operating sewer pipes, represented by the 16 data from this investigation,
- observations obtained in cleaned operating sewer pipes, represented by the 74 data from Parkhurst and Pomeroy (1972), and
- observations obtained in a clean laboratory pipe, represented by the 50 data from Taghizadeh-Nasser (1986).

Coefficients of roughness were estimated by the author for all the observations of Parkhurst and Pomeroy (1972) and Taghizadeh-Nasser (1986). In general, both sets of observations were obtained in pipes with coefficients of roughness typically less than 0.1 mm. Except for the observations obtained under high-flow conditions (Test Nos. 16 to 20), this is clearly under the level of roughness found for the pipes from this investigation (Table 2). The observations of Parkhurst and Pomeroy (1972) include seven observations obtained in a sewer with an atypical high slope resulting in relatively high K_La values. In some of the following considerations these seven observations will be excluded.

In Figure 5, the five empirical models were applied on the complete series of observations. Not surprisingly, the Parkhurst-Pomeroy equation looks like the most promising equation. Equations 10, 11, and 14 still overestimate the K_La value significantly in the low range of K_La observations. However, the Tsivoglou-Neal equation now seems more reasonable on an average, represented by a residual average of 0.27 in Table 5, but compared with the Pomeroy-Parkhurst equation, the scattering around the centerline, represented by the standard deviation, is clearly higher.

When the plots in Figure 5 (especially 5a, 5c, and 5e) are observed, it is seen, that the points representing the Taghizadeh-Nasser laboratory observations differ from the operating sewer observations. For observed K_La values from approximately 2 to 5 h^{-1} the corresponding estimates from the equations seem lower for the laboratory observations than for those of the operating sewers. As the observed K_La values increase, the laboratory observations approach the seven relatively high field observations of Parkhurst and Pomeroy. From analysis of the published data it is not likely that this systematic deviation could be attributed to systematic differences of system parameters (such as diameter, slope, and wall roughness) or hydraulic parameters (such as flow, velocity, depth, and hydraulic radius). However, the deviation may be due to a systematic difference of the alpha factor, representing the difference between tap water and wastewater tests. Note that the Parkhurst and Pomeroy equation (including an unknown value of the alpha factor) was applied unchanged on the tap water observations.

It is well known that contaminants under certain conditions significantly affect gas transfer (Gilbert, 1979). If the discrepancy

of the Taghizadeh-Nasser data, obtained from tap water tests, should be explained by a systematic variation in the alpha factor, then its value should be approximately 0.3 at a K_La level of 5 h^{-1} , and this value should increase toward unity or higher as the K_La level increases. This type of variation corresponds to the following concept presented by Gilbert (1979) and illustrated on Figure 6. According to Gilbert an interfacial film of surfactants should have little effect under quiescent conditions because gas transfer is governed by molecular diffusion also in the bulk phase; at intermediate levels of turbulence the surfactants may increase the interfacial resistance and thereby decrease the transfer rate; at high mixing levels no interfacial barrier is permitted to establish, and bubbles are formed, thus alpha may return to or even get greater than unity. On the other hand, there is hardly any systematic tendency such as the one shown in Figure 6 in the wide range of published alpha factor observations.

The laboratory experiments shown in Table 3 and 4 do not show any convincing deviation of alpha from unity for tests made at a K_La level typical for several of the real sewer observations. Of course, because alpha certainly also depends on system properties, values of alpha measured in a laboratory vessel cannot really be applied to a sewer pipe. However, it should be expected that if alpha was in the order of 0.3 for the field observations with a K_La value less than 5 h^{-1} , at least some effect of impurities would have been detected in the case of the laboratory tests. This probably indicates that at least a part of the systematic deviation of the Taghizadeh-Nasser observations could be explained by a systematic error introduced by the laboratory equipment or the laboratory procedure. Therefore, it was decided to exclude the Taghizadeh-Nasser data from further considerations.

The equations of Owens *et al.* (1964) and Krenkel and Orlob (1962) were developed for data obtained in brooks and rivers, and a rectangular laboratory channel, respectively. In the case of Figure 5 they are both good examples of equations used outside their range of applicability. In spite of the fact that most of the observed sewer velocities, depths, and slopes can be considered within the range of those used for derivation of these equations, sewer observations apparently cannot be compared with neither river nor rectangular channel observations.

Also, the Tsivoglou and Neal equation was derived from river observations. As previously mentioned, it seems more reasonable on an average. However, there is a remarkable difference between the points representing this investigation and the points representing the Parkhurst-Pomeroy observations. This probably reveals that the equation is incapable of taking the effect of roughness into account. This may also be the case of the Owens *et al.* equation. Furthermore, the model of Tsivoglou and Neal is atypical, compared with the other empirical equations, as for a given sewer pipe (Figure 1) it predicts increasing K_La values for increasing flow conditions. Consequently, it may succeed in estimating the level of gas transfer, but it fails when it comes to prediction of a variation due to varying flow conditions. This is primarily due to the missing dependency of the hydraulic mean depth.

It is likely that turbulent kinetic energy in a circular sewer pipe is much more uniformly distributed than in a rectangular pilot channel, for example, Krenkel and Orlob (1962), that contains sharp-angled edges, or in brooks and rivers of various shapes, for example, Owens *et al.* (1964) and Tsivoglou and Neal (1976). Energy distribution can have a significant effect on

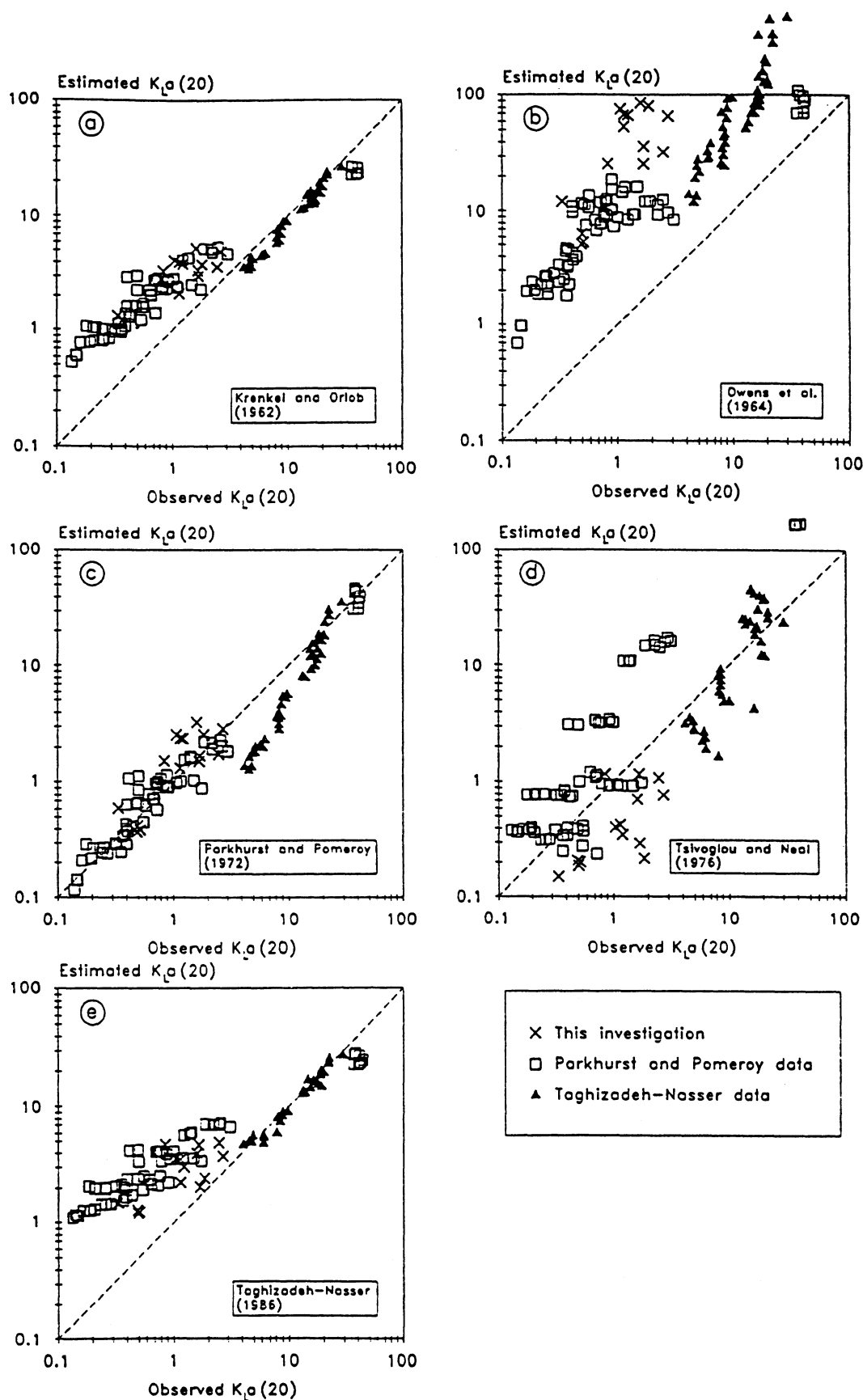


Figure 5—Plot of predicted versus observed $K_L a$ values (h^{-1}) based on the complete series of observations.

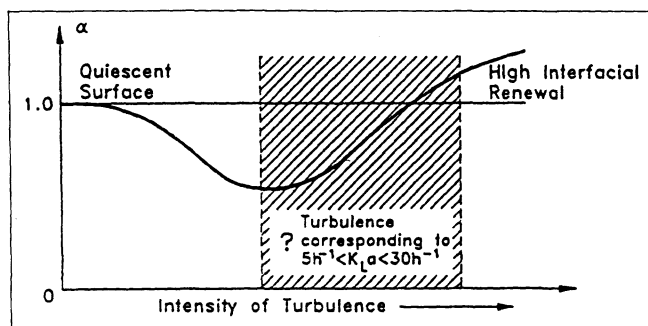


Figure 6—Relationship between alpha and turbulence at a given surfactant concentration as outlined in Gilbert (1979).

$K_L a$, and differences in the geometries of experimental systems will affect energy distribution.

Taghizadeh-Nasser (1986) presented Equation 12 as an example among 19 different predictive equations, all with correlation coefficients greater than 0.93, when applied on his laboratory observations. However, when observations are spread over a wide range, the correlation coefficient is hardly a sufficient evaluation criterion. When the equation is applied on the operating sewer data, it clearly fails in predicting the oxygen-transfer coefficient, especially for the low observations of $K_L a$. This deviation can hardly be explained by alpha alone.

The Parkhurst and Pomeroy equation. Because the majority of the operating sewer observations originate from the work of Parkhurst and Pomeroy (1972), their equation, as expected, turns out to be the best. However, the observations originating from this investigation verify the validity of the Parkhurst-Pomeroy equation reasonably when applied on observations outside the base of development. In this connection, it should be noticed that contrary to the Parkhurst-Pomeroy investigation, this investigation was conducted on noncleaned pipes, with a high coefficient of roughness. Furthermore, in this investigation the direct tracer method was used, contrary to the mass-balance method used by Parkhurst and Pomeroy. Finally, though no wastewater characteristics were presented in Parkhurst and Pomeroy (1972), it is tempting to expect that a variability of wastewater characteristics are covered through the two series of operating sewer observations on $K_L a$. Therefore, if wastewater characteristics have a significant influence on $K_L a$, expressed through alpha, then a greater scatter in the points plotted in Figure 5(e) probably should have been expected.

Even though the statistical measures are fairly good for the Parkhurst and Pomeroy equation, the residual average, being 0.08 for the complete series of operating sewer data, still indicates that the equation may tend to overestimate $K_L a$ by approximately 8%. Therefore it was decided to make a nonlinear regression (by a Marquardt routine) on the 90 observations from operating sewers, using the following transformation of Equation 12:

$$\ln(d_m \cdot K_L a) = \ln(a) + \ln(1 + b F^2) + c \ln(s \cdot u) \quad (21)$$

The coefficients (a, b, and c) in Equation 21 were found to be 0.45, 0.39, and 0.29, respectively. If Equation 12, given these coefficients, is applied on the 90 observations SE is found to be 0.295, and RA to be 0.0005. Note that the statistical measures

obtained differ slightly from the optimum because the coefficients obtained by regression have been rounded.

The value 0.375 of the coefficient c used by Parkhurst and Pomeroy, originates from the theoretical considerations of Dobbins (1964) based on surface renewal theory. Using the Kolmogoroff scale of turbulence among other, Dobbins (1964) assumed that the surface renewal rate was related to $E^{3/4}$, E being the dissipation of energy per unit of mass and unit of time, given by $E = s \cdot u \cdot g$. Thus, if the value of r is sufficiently high, for the coth term in Equation 7 to be approximately unity, then K_L should be related to $E^{3/8}$. Consequently, a new nonlinear regression was made, giving the coefficient c a fixed value of 0.375. Thereby, the following version of Equation 12 was obtained:

$$K_L a(20) = 0.86 (1 + 0.20 F^2)(s \cdot u)^{3/8} d_m^{-1} \quad (22)$$

If Equation 22 is applied on the 90 observations, SE is found to be 0.298. Furthermore, RA is found to be -0.004, indicating that the equation tends to underestimate $K_L a$ in the order of 0.4%. Compared with the general scatter of the estimates, this deviation from zero is insignificant. Because the statistical measures, obtained for Equation 22, are found only slightly poorer, than those obtained when c was allowed to vary by the regression procedure, there are no reasons not to follow the theoretical considerations of Dobbins (1964). Therefore, Equation 22 is recommended as a slightly improved version of the Parkhurst and Pomeroy equation. If applied on the 83 data series, the statistical measures for Equation 22 are almost identical with those found for the complete operating sewer series.

As seen in Figure 7, Equation 22 hardly improves the scatter of the points, also represented by the SE. A reason for that could be that one or more parameters, such as the roughness coefficient, is missing in the equation. By a trial and error procedure the scatter might have been reduced. However, any small improvement in the SE does not necessarily mean that a more reliable equation is found. Furthermore, one should not forget that gas-transfer coefficient measurements include a considerable scatter. Small alpha variations could also be the reason for a part of the observed scatter.

The effect caused by flows from side connections such as service laterals on the gas-transfer coefficient in the main gravity sewer pipe is one condition that needs clarification. On the basis of the statistical measures in Table 5 and visual observation of Figure 7, it seems as if the observed values obtained in this investigation in general tend to be lower than those of Parkhurst and Pomeroy. There were no side connections to the pipes chosen for this investigation. Therefore, it is possible that side connections have an increasing effect on the gas-transfer coefficient. Unfortunately, there is no information to be found in Parkhurst and Pomeroy (1972) on whether side connections were present.

Estimating oxygen transfer. When estimating the air-to-water oxygen transfer in gravity sewers, not only the $K_L a$ value should be found. Equation 9 shows that also values of alpha, beta, and C_s are needed. When using Equation 12 and Equation 22, alpha is included in the estimated $K_L a$ value. However, it is not clear to what extent the included value of alpha differs from unity. In Parkhurst and Pomeroy (1972) it was believed "that alpha for stream aeration of sewage is between 0.3 and 0.4." On the other hand, if the conducted laboratory investigations are representative of gravity sewer conditions, the alpha factor apparently is close to unity.

In this investigation no effort was made to estimate the value

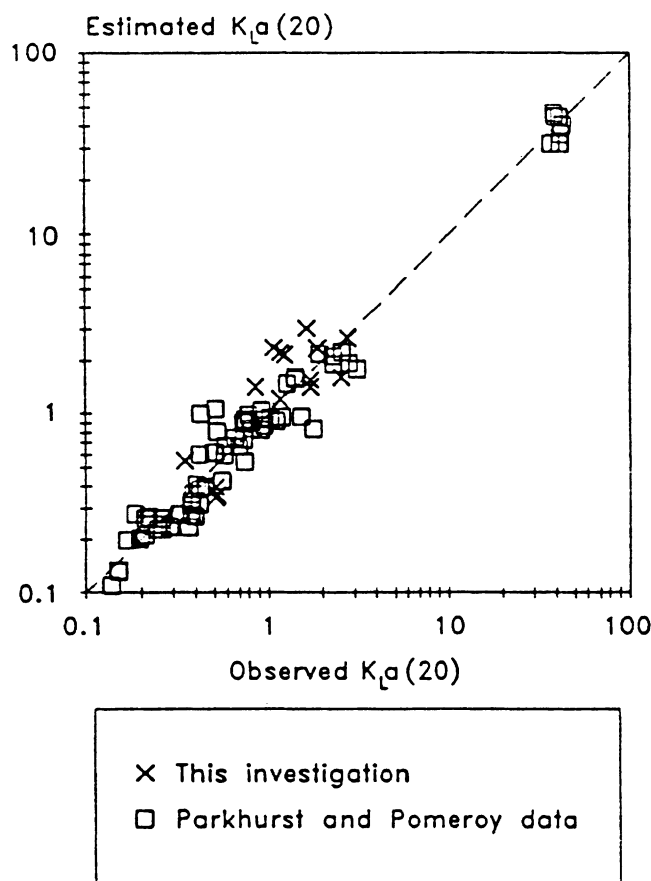


Figure 7—Plot of predicted versus observed K_La values (h^{-1}) based on the improved version (Equation 22) of the Parkhurst and Pomeroy (1972) equation.

of beta. As stated in WPCF/ASCE (1988), values of beta are expected to range between 0.8 and 1.0 in the case of oxygen solubility in wastewater, and for municipal wastewater beta is generally close to 1.0. Therefore, it is normally accepted to use a value of 0.95. The possible error introduced by using this value of beta is most probably less than 5% to 10%. Compared with the scatter in the estimation of K_La this error is of minor importance.

A more significant error can be introduced in the estimation of oxygen transfer in gravity sewers if tabulated values for oxygen solubility in tap water are used without correction for the partial pressure of oxygen in the sewer atmosphere. Dissolved oxygen is continuously being removed from the flowing wastewater because of microbial and chemical processes. Therefore, if sewer ventilation is insufficient, the air-to-water interface transport of oxygen may result in a substantial reduction of the oxygen partial pressure in the sewer atmosphere. As previously mentioned, partial pressures observed in this investigation were as low as 17.3%. In many cases probably even lower values will be found. It is therefore obvious that some attention must be drawn to the estimation of partial pressures in the sewer atmosphere when modeling oxygen transfer in gravity sewers.

Conclusions

Among five evaluated empirical models for the air-to-water oxygen-transfer coefficient, the equation of Parkhurst and Pom-

eroy (1972) was found to be the most precise when applied on operating gravity sewers. The evaluation was based primarily on the 16 observations obtained in this investigation and secondarily on the 74 observations presented by Parkhurst and Pomeroy. The two series of data represent observations obtained in noncleaned and cleaned operating sewer pipes, respectively. Thereby conditions of both high and low wall roughness were included in the evaluation of the empirical models.

By nonlinear regression using the complete series of operating sewer data, an improved version of the Parkhurst and Pomeroy equation was derived. Because the original equation probably overestimated by approximately 8% on average, it is recommended that Equation 22 be used in the future. However, it should be recognized that the deviations between the Parkhurst and Pomeroy equation and the operating sewer observations probably are well within the experimental error. Furthermore, if normal sewer design practice is taken into consideration, an overestimation by 8% is insignificant.

Like the original equation of Parkhurst and Pomeroy, Equation 22 includes the effects of impurities. The laboratory investigations carried out on alpha showed no deviation from unity, except when a substantial amount of foam was present.

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