TRANSFORMATION OF WASTEWATER IN SEWER SYSTEMS – A REVIEW

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ABSTRACT

Many chemical, physical and biological transformations of wastewater in sewer systems may take place and cause significant changes in the composition during transportation. This paper focuses on the methods for measuring changes in wastewater composition (organic matter, dissolved oxygen, nitrogen and sulfur compounds), characteristics for changes in wastewater composition and empirical equations for oxygen uptake rates, sulfide production rates and BOD removal rates in both pressure mains and gravity sewers. Simple and more complicated biofilm models are evaluated for use in sewer systems, but so far no suitable advanced model for sewer biofilms exists, due to a lack of quantitative information about the transformations in such high-loaded multispecies biofilms.

KEYWORDS

Wastewater composition; pressure mains; gravity sewers; biofilm; oxygen uptake rates; sulfide production rates.

INTRODUCTION

During transportation of wastewater in a sewer system significant changes in both quantity and quality of the organic matter and the electron acceptors occur. In some cases large BOD removal rates have been observed and this ability of the sewer system to self-purification has been discussed (Pomeroy and Parkhurst, 1973, Boon et al., 1977). In other cases, especially in connection with full flowing pipes (pressure mains) in which anaerobic processes dominate (e.g. sulfide production), odour problems, health aspects and corrosive effects on metal and concrete are all well-known phenomena (USEPA, 1974 and 1985). Due to increased demand for nutrient removal of the wastewater, increasing efforts are made to characterize the wastewater for optimal use in biological nitrogen and phosphorus removal plants (e.g. Sollfrank and Gujer, 1991). In this connection it is interesting that the hydraulic retention time of the wastewater in many sewer systems is of the same order of magnitude as in the treatment plant. Increased knowledge of the composition of the wastewater based on its fate in the sewer system is therefore needed.

The design principles of sewer systems as well as strength and composition of the wastewater and climatic conditions vary throughout the world. In this paper the characteristics of the sewer system are discussed by highlighting experience and practice in Denmark. Sanitary sewers for transportation of the dry as well as the wet-weather flow of mainly domestic wastewater will be dealt with in this connection. Storm sewers will not be dealt with. The aim of this study was to describe the present knowledge and show some results

concerning the microbial processes taking place in bulk water, sediment and biofilm in sewer systems. An increased knowledge about these processes will improve our ability to:

-      predict the quantity and quality of the organic fractions in the wastewater as input to the treatment plant;
-      predict and control the production of noxious odours, toxic and corroding compounds (e.g. sulfide);
-      predict the loads and effects of overflows to receiving waters in combined sewers;
-      model and simulate the processes in sewer systems and take this knowledge into account in designing and operating new sewer systems.

WASTEWATER COMPOSITION

Usually domestic wastewater has a very complex composition of both inorganic and organic compounds. In this paper special attention is devoted to the organic part and some inorganic compounds which are important in the biological processes, mainly oxygen, nitrate, ammonium, sulfate and sulfide. The composition of "young" wastewater with an age of only minutes or a few hours may be quite different from wastewater which has been under transportation for 20 hours or more. This is due mainly to microbial growth and respiration in both bulk water and in biofilms, solubilization, enzymatic hydrolysis of macromolecules and hydraulic shear forces. Also sedimentation and resuspension may be important. The significance of these factors is strongly affected by the design of the sewer system, e.g. gravity or pressure pipes. The composition of the wastewater is normally investigated as inflow to a treatment plant, and several authors have stated that the composition besides diurnal and seasonal variations strongly depends on the design of the sewer system and the residence time of the wastewater (USEPA, 1974, Levine et al., 1985), but very few measurements have documented this observation.

The organic matter in wastewater has traditionally only been characterized by total COD and BOD. On a few occasions a more detailed description of the individual constituents, the biodegradability or the particle size distribution has been focused on.

BOD, COD and TOC can be divided into constituents such as lipids, carbohydrate, proteins, organic acids and other substances. Relatively few measurements are published but some examples are shown in Table 1. In a few early studies several individual components have been identified, for instance individual lipids and organic acids, carbohydrates and amino acids (Heukelekian and Balmat, 1959, Painter and Viney, 1959, Hunter and Heukelekian, 1965).

Table 1. Examples of the composition of domestic wastewater. (Percentage of total COD)

Component	Henze, 1982	Tanaka et al., 1991	Narkis et al., 1980*			
			**		***	
Total COD (g/m^3)	530	259	394	(350-443)	813	(600-1180)
Soluble COD	40		57	(50-70)	36	(33-38)
Carbohydrate	12	6	9	(2-20)	6	(2-9)
Proteins	8	12	27	(18-36)	14	(7-22)
Lipids	10	19				
Surfactants	5	6	8	(7-10)	4	(3-5)
Fatty acids	19	3	10	(6-12)	14	(8-18)
Humic acids		2				
Other organics	46	49				

* average and range for 3 samples for each type of wastewater
** young wastewater (10-60 minutes), *** old wastewater (6-12 hours).

Prior to analysis the wastewater is often fractionated into soluble and particulate matter, defined by filtration with a pore size of 1.2 (GF/C) μm or in some cases 1.8 (GF/A), 0.45 or 0.2 μm. Most of the BOD or COD is usually found in the particulate fraction. A detailed study of particle size distribution of settled domestic

wastewater has been performed by Levine et al., (1985) at two US locations. It was found that approximately 63-70% of the organic material was associated with particles larger than 0.1 μm. Using gel filtration chromatography they found very little organic matter in the size range between 0.01 and 0.1 μm, while a peak appeared in the molecular mass range less than 20,000 amu (atomic mass units). Based on statistical analysis of the particle size distribution for 34-42 samples of sewer sediments from 11 locations in the UK, Binnie and Partners (1986) concluded that the sewer sediment is relatively coarse with little material smaller than fine sand (60 μm) and with 18% of organic matter as an average value. During high flow periods, e.g. runoff events, these sediments may contribute considerably to the transportation of material in the sewer. Verbanck et al., (1990) investigated sewer sediments as well as suspended solids under transportation in a large sewer main in Brussels. Based on 22 series of measurements it was concluded that the sewer sediments during runoff events contributed with mainly small particles (10-100 μm) to the suspended solids in the water phase. The amount of organic matter in the sediments was typically 20-40%. To the authors knowledge no measurements of changes in particle size distribution during transportation in sewer systems have ever been published.

Another method of characterizing the wastewater is to estimate the biodegradability of particulate and soluble matter based on oxygen uptake rate measurements (e.g. Marais and Ekama, 1976, Sollfrank and Gujer, 1991). So far this method has mainly been used for incoming wastewater to treatment plants because the characterization is closely connected to the modelling of the activated sludge processes.

CHARACTERISTICS OF GRAVITY AND PRESSURE MAINS

In Denmark combined sewers serve approximately 55% of the urban catchment area, whereas the remaining 45% have separate sewers, which have been constructed during the last 35-40 years. Gravity sewers are predominant, but pressure mains connected with centralized wastewater treatment plants have been more common during the last 10-15 years. Accordingly, the number of pumping stations with wet wells has increased. Furthermore, with the rising concern for receiving water protection during wet weather periods, the number of detention basins in the combined sewer systems has increased. Generally, the trend during the last decade has led to an increase in residence time of the wastewater in the sewers and to situations where anaerobic conditions prevail.

Considering wastewater transformations, a sewer system and especially a combined one is characterized as highly dynamic. Fluctuations in flow and concentrations over day and night are well documented, and during a runoff event the flow may increase with a factor 10 to 100. Such conditions may affect sedimentation and resuspension of the sewer sediments, scouring of the biofilm and changes in the redox conditions. These phenomena may result in delayed effects in the sewers with duration of several days. *However, it is not the objective of this paper to discuss extreme wet-weather impacts, but it should be kept* in mind that they frequently take place.

It is interesting to mention that in Tel-Aviv, Israel, a 37 km loop-shaped sewerage system is used as an aerobic step-fed plug-flow reactor for organic matter removal (Green et al., 1985). The reactor operates by circulating activated sludge biomass and by the injection of air or oxygen.

Gravity sewers

The upper part of a gravity sewer system has normally a slope of approximately 0.5-1.5%. This slope gradually decreases to about 0.1-0.5% with low values typical for intercepting sewers. In order to maintain self-purification in the sewers, a design flow velocity around 1 m/s or a shear stress around 2-4 N/m^2 is typical. Average flow velocities vary typically between 0.5 and 1 m/s, and in practically all municipalities it is necessary to flush certain sewers in order to compensate for too low design flow and prevent flooding.

In Danish gravity sewer systems the oxygen concentrations are typically 1-4 mg/l. Therefore, the oxygen consumption and exchange processes in the water phase, in the biofilm and at the sediment/water and water/sewer atmosphere surfaces play an important role for changes in the wastewater composition.

The biofilm in a gravity sewer is typically 1-3 mm thick. The thickness of the organic-rich and cohesive sediment may vary considerably, but several centimetres is common and anaerobic conditions - except for the surface - exist. Sediments up to 80 cm are, however, reported for Japan in large sewer trunks (Mori et al., 1991). Very little is known about the dynamics of sedimentation and resuspension although the contribution to the concentration of suspended solids is observed during runoff events. Also external impacts may affect the wastewater composition. Leaks in sewer pipes are common in Denmark and infiltration or exfiltration - depending on the groundwater level - is a potential threat. In wet periods infiltration may account for 20-30% or more of the daily flow. Particularly sewers constructed during the period 1930-1950 need rehabilitation. Sewers constructed today are not expected to result in any infiltration. Changes in wastewater composition may also take place due to volatilization of gases into the sewer atmosphere.

The annual variation in wastewater temperature is typically 8-18°C. In a combined system in winter periods the temperature may drop considerably due to inflow of rain or melted snow. Industrial wastewater can increase the temperature, but according to current regulations discharge temperature should be below 30-35°C.

Pressure mains

Pressure mains are permanently full flowing sewers which operate intermittently. During pumping periods flow velocities vary between approximately 0.8 and 1.3 m/s. Dependent on pump capacity and actual flow, these maximum flow velocities may be reduced with a factor 2-20 in order to obtain a mean flow velocity. Especially in combined sewered catchments low mean velocities are usually the case. Therefore, compared with gravity sewers increased residence time may be observed in pressure mains.

Except for the initial part of a pressure main, strict anaerobic conditions may exist. Sulfide production and an attending consumption of biodegradable organics may take place, predominantly in the 0.1-0.5 mm thick biofilms at the wall. Sediment accumulation is probably possible, but real operational problems have not been reported. Pressure below 1 atmosphere may occur in parts of a pressure main. Therefore, the solubility of gases e.g. N_2 in the wastewater may be exceeded and ventilation needed in order to maintain proper operational conditions of the pressure main.

METHODS TO EVALUATE CHANGES IN THE ORGANIC MATTER
COMPOSITION IN WASTEWATER

It is possible to sample and record changes directly in the wastewater of sewer pipes measuring at upstream and downstream locations taking the residence time into account. Alternatively, some investigators have taken water samples from different places in the sewer system and performed laboratory activity test. In some cases investigations are done using biofilm reactors in laboratory-scale. The methods used are as follows,

1)   Measurements of changes in the total amount of organic matter using total BOD, COD or TOC. Boon et al. (1977) used changes in total BOD as a measurement for removed organic matter during transportation in gravity and pressure mains.
2)   Measurements of changes in specific organic pools or in the concentration of specific compounds (e.g. volatile fatty acids). This method has been used by Narkis et al. (1980).
3)   Measurements of potential hydrolytic and respiration activity by enzyme activity tests. Obst and Holzapfel-Pschorn (1986) measured enzyme activity (esterase and electron transport system) in wastewater.
4)   Measurements of consumption of electron acceptors (e.g. oxygen, nitrate, sulfate) or the appearance of products (e.g. sulfide) and then estimate the equivalent amount of organic matter consumed.

TRANSFORMATIONS IN GRAVITY SEWERS AND PRESSURE MAINS

Oxygen: The concentration of dissolved oxygen in the bulk water in gravity sewers is a result of a balance between microbial consumption processes and reaeration from the sewer atmosphere. A simple oxygen balance in flowing wastewater may be given as:

$$dS_O/dt = K_L a(S_{O,m} - S_O) - (r_w + r_b + r_s)$$

The first term describes the input of oxygen over the interface between the sewer atmosphere and the flowing wastewater. $K_L a$ is the reaeration coefficient (1/h), $S_{O,m}$ and S_O saturated and actual oxygen concentration in wastewater, respectively (g/m^3). The oxygen consumption rates in the bulkwater (r_w), the biofilm (r_b) and the sediment (r_s) are given in units of g/m^3.h.

The measurements of reaeration rates in sewers based on a dissolved oxygen balance have been presented by Parkhurst and Pomeroy (1972). These authors have also proposed an empirical equation for the prediction of reaeration rates in sewers. Jensen and Hvitved-Jacobsen (1991) have used a radio-tracer method for direct measurement of reaeration rates and they found reasonable agreement with the formula of Pomeroy and Parkhurst. The reaeration coefficient at 20°C can be represented by:

$$K_L a = 0.96(1 + 0.17(u^2/gd_m))(su)^{3/8} d_m$$

u is the velocity (m/s), g the gravitational acceleration (m/s^2), d_m the hydraulic mean depth (m) and s the slope of the sewer (m/m). The reaeration coefficient is temperature dependent which is discussed by Jensen (1990).

Bulk water oxygen consumption rates in gravity sewers do not differ from the rates in pressure mains. Oxygen addition to pressure mains is often used as an effective control strategy against sulfide formation. Therefore, extensive knowledge has evolved on the oxygen consumption in bulk water and biofilm (USEPA, 1974, Boon and Lister, 1975). They found that the rates in bulk water range generally from 2-3 to 20 mg O_2/l.h depending on the age of the wastewater, with an average of 15 mg O_2/l at 15°C . At 20°C they estimated the oxygen uptake rate for wastewater with an age of 1, 2 or higher than 3 hours to be 5, 10 and 15 mg O_2/l.h, respectively.

Oxygen consumption of the biofilm in both pressure mains and gravity sewers is due to direct microbial oxidation of organic matter and oxidation of reduced substances (e.g. sulfide) produced from the deeper part of the biofilm. This rate is quantified by Pomeroy and Parkhurst (1973) by an empirical equation based on 48 tests in 8 sewers (Table 2, equation 5). They found that the rate was directly proportional with the oxygen concentration, but also slope of sewer and water velocity were important. Values as high as 1.4 gO_2/m^2.h were measured at water velocities of 2.2 m/s, slope 0.0097 m/m and temperature 23°C. Since oxygen supply from an air phase is absent in pressure mains, the oxygen in the wastewater is usually depleted within a few minutes. By injecting oxygen for sulfide control Boon and Lister (1975) found the contribution from the biofilm to be relatively constant with 0.7 gO_2/m^2.h as an average value at 15°C. The oxygen consumption of the sewer sediment has not been investigated until now. However, from studies of three rivers with significant wastewater loadings Hickey (1988) found an average benthic oxygen uptake rate of 0.75 gO_2/m^2.h; a value as high as 2.9 gO_2/m^2.h was recorded at a water velocity of 0.25 m/s, a DO concentration of 7 mg/l and a temperature of 22°C. Boyle and Scott (1984) found oxygen uptake rates around 0.4 gO_2/m^2.h at 20°C in a river receiving papermill effluent.

To illustrate the effect in a gravity sewer of the equations mentioned above, values of $K_L a, u, S_O$ and r_a (biofilm surface rate, gO_2/h^2.h) are plotted against the flow Q for a steady state situation (Figure 1). As an example a gravity sewer with a diameter of 500 mm and a slope s = 0.003 m/m is selected. The sewer is assumed without sediment, but with an active biofilm on the wetted perimeter. Bulk water respiration rate (r_w) is assumed to be 5 mg/l.h at 10°C. The illustrated sewer is full flowing at approximately 215 l/s (775 m^3/h). In case of small flow rates the dissolved oxygen concentration is in the range of 2-4 mg/l. When the

flow increases the oxygen concentration decreases significantly, and at around 90 l/s (325 m^3/h) no dissolved oxygen appears in the flowing wastewater. This is mainly due to the fact that the input from the reaeration process at this level becomes less than the rate of the oxygen consumption in the wastewater. In Denmark the typical flow (Q) is low, usually 10-20% of the full capacity even in separate sewers. Combined with the common low temperatures the oxygen concentration in the bulk water is normally high enough to prevent sulfide build-up.
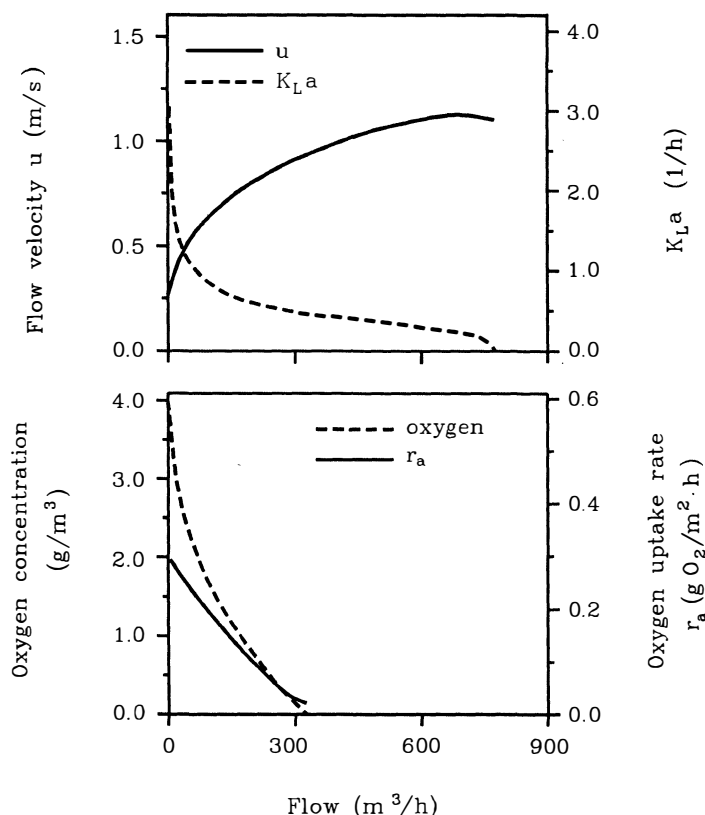

Figure 1. Calculated reaeration coefficient, water velocity, oxygen concentration in bulk water and biofilm oxygen uptake rate in a gravity sewer at various water flows. Temperature 10°C.

Nitrogen compounds: High concentrations of nitrate are usually not present in sewer systems. If present it is denitrified deep in the biofilm in the presence of oxygen or in both bulk water and biofilm when oxygen is depleted. The permanent presence of oxygen in gravity sewers increases the possibility of some nitrification in the biofilm. However, nitrifying activity is usually not significant in biofilms at high organic loadings (e.g. Harremoës, 1982, Chen et al., 1989). Therefore, in most sewer systems denitrification probably only occurs if nitrate or nitrite is discharged or infiltrated into the sewer. In some cases nitrate has been added to the wastewater in pressure mains instead of oxygen to suppress sulfate reduction (Jacobsson, 1981). Henze (1985) compared denitrification rates with the oxygen uptake rates in bulk water of 5 types of domestic

wastewater. He found that the denitrifying activity ranged from 0-80% of the oxygen respiration rate (measured as electron equivalents). Typical values were 25-55%. Large variations were found between different types of wastewater, while the variations for a certain type of wastewater were smaller. Ammonification during fermentation and respiration of organic matter is likely to happen, but we have not been able to find any quantitative studies from sewer systems in the literature.

Sulfur compounds: Sulfide production from sulfate reduction in gravity sewers takes place mainly in slow-flowing, large pipes with insufficient reaeration at relatively high temperatures, and in pressure mains when the residence time of the wastewater is more than 1-2 hours. Sulfide in bulk water in gravity sewers in Denmark is rare, probably due to the low temperatures. On the other hand, in Danish pressure mains sulfide build-up during transportation is typically up to 10 mgS/l. In other countries with longer pressure mains or higher temperatures significantly higher concentrations are reported (e.g. Thistlethwayte, 1972, Pomeroy and Parkhurst, 1977).

Sulfide production is a big problem due to noxious odour, health aspects and corrosive effects on metal and concrete in the sewer system and at the treatment plant. Sulfide in the wastewater may also affect the biological processes in the wastewater treatment, e.g. bulking problems (Strom and Jenkins, 1984) and it may be toxic for fish in streams affected by overflow events. In cases with sulfide control, e.g. injection of oxygen, nitrate, hydrogen peroxide, chlorine, iron salts or lime, this control can also affect the preceding treatment.

The sulfate reduction in sewer systems takes place mainly in the biofilm, although some activity may occur in bulk water due to detached biofilm particles (USEPA, 1974). The most important factors determining sulfate reduction rates are the presence of sulfate or other oxidized sulfur compounds, the quantity and the quality of the organic matter and the temperature.

Table 2. Empirical equations for forecasting sulfide production rates in pressure mains (1-4) and oxygen consumption in gravity sewers and pressure mains (5). Unit for r_a is $g/m^2.h$.*

Equation no.		Reference
1	$r_a = 0.5 \times 10^{-3} u C_{BOD}^{0.8} S_{SO_4}^{0.4} 1.139^{(T-20)}$	Thistlethwayte, 1972
2	$r_a = 0.228 \times 10^{-3} C_{COD} 1.07^{(T-20)}$	Boon and Lister, 1975
3	$r_a = 1 \times 10^{-3} C_{BOD} 1.07^{(T-20)}$	Pomeroy and Parkhurst, 1977
4	$r_a = k** \times 10^{-3} (S_{COD} - 50)^{0.5} 1.07^{(T-20)}$	Nielsen and Hvitved-Jacobsen, 1988
5	$r_a = 5.3 S_O(su)^{0.5}$	USEPA, 1974

* Can be expressed as $g/m^3.h$ by dividing with hydraulic radius (V/A)
** k = 1.5 for Danish domestic wastewater, 3 for wastewater with some contribution from food industry and 6 for easily degradable wastewater from food industry, (see Figure 2, A, B and C).

Several empirical equations are developed to predict the sulfide production rate from the biofilm surface in pressure mains (in $gS/m^2.h$, Table 2). Equations 2, 3 and 4 assume that the sulfate concentration is high and nonlimiting, which is usually the case when the concentration is above 4-5 $mgSO_4$-S/l (Nielsen and Hvitved-Jacobsen, 1988). Typical concentrations in Danish wastewater are usually above this value. Equations 1 and 2 estimate maximum possible rates and they are a conservative result of measurements on many pressure mains. Equation 4 (shown in Figure 2) is useful for predicting variations in the sulfide build-up in a pressure main when the actual type of wastewater is known. It has been shown suitable for estimating diurnal and seasonal variations in the sulfide production rate in many Danish combined or separate pressure mains (Hvitved-Jacobsen et al., 1988 and Hvitved-Jacobsen and Nielsen, unpublished results). Sulfur compounds such as sulfite and thiosulfate are also reduced to sulfide if present in wastewater. Nielsen (1991) showed that these compounds are reduced instead of sulfate by the sulfate reducing bacteria and

often result in increased sulfide production rates. On the other hand organic sulfur compounds usually seem to be insignificant sources for sulfide production.
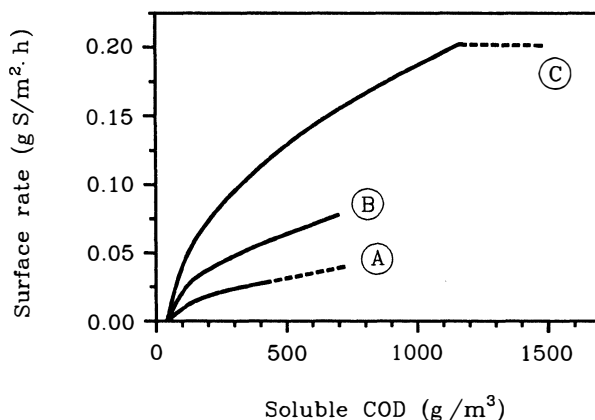

Figure 2. Prediction of sulfide formation in pressure mains (Hvitved-Jacobsen et al., 1988). See Table 2.

In gravity sewers there is usually no sulfide present in bulk water when the oxygen concentration is above 0.2-0.5 mg/l (USEPA, 1974). When the oxygen concentration is lower because of high oxygen consumption rates or low reaeration rates, the empirical equation 3 which is shown in Table 2 can be used successfully, although the rate constant is usually lower than 1×10^{-3} (USEPA, 1974). Moreover, the other equations which are normally used for pressure mains seem to be in accordance with real situations. An estimation of the sulfide concentration in bulk water, however, has to take loss to the atmosphere and chemical and biological oxidation in bulk water into account (Pomeroy and Parkhurst, 1977, Tchobanoglous, 1981, Wilmot et al., 1988).

The flow velocity in pressure mains, which is important for the oxygen consumption rate of the biofilm, may also affect the sulfide production rate. At very low average velocities a shortage of nutrients transported to the biofilm may occur, while very high velocities may diminish the biofilm thickness and activity (USEPA, 1985, Holder and Hauser, 1987).

Organic matter: Significant amounts of organic matter may be removed during transportation in sewers. Pomeroy and Parkhurst (1973) assume that the high oxygen consumption rate mentioned above caused significant self-purification of the organic matter. However, only few measurements of the BOD change are presented. Boon et al. (1977) found by injecting oxygen into a force main, that generally 1 gramme BOD was removed per gramme of oxygen consumed. The removal was between 0.7 gBOD per gO_2 during winter (11-14°C) and 1.4 gBOD per gO_2 during summer (20-25°C).

Only a few studies have documented the suggested changes in the organic composition during transportation in anaerobic pressure mains. Narkis et al., (1980), Table 1, analysed the composition of domestic wastewater from two sewer systems in Israel, one with a low age (10-60 minutes), and one with a high age (6-12 hours) in a pressure main. The main difference was an increased amount of volatile fatty acids (VFA), especially acetate and some propionate in the system with long residence time. They also analysed the changes in VFA composition in wastewater stored anaerobically for 24 hours. They found increases in the concentration of acetate from 0-48 mg/l (0-110%) at 5°C and 8-105 mg/l (21-295%) at 25°C. Also the concentration of propionic acid was found to increase, but the concentrations were generally much lower.

These observations are in accordance with experience from other anaerobic systems, e.g. anaerobic digestion, in which fatty acids are produced during hydrolysis and fermentation.
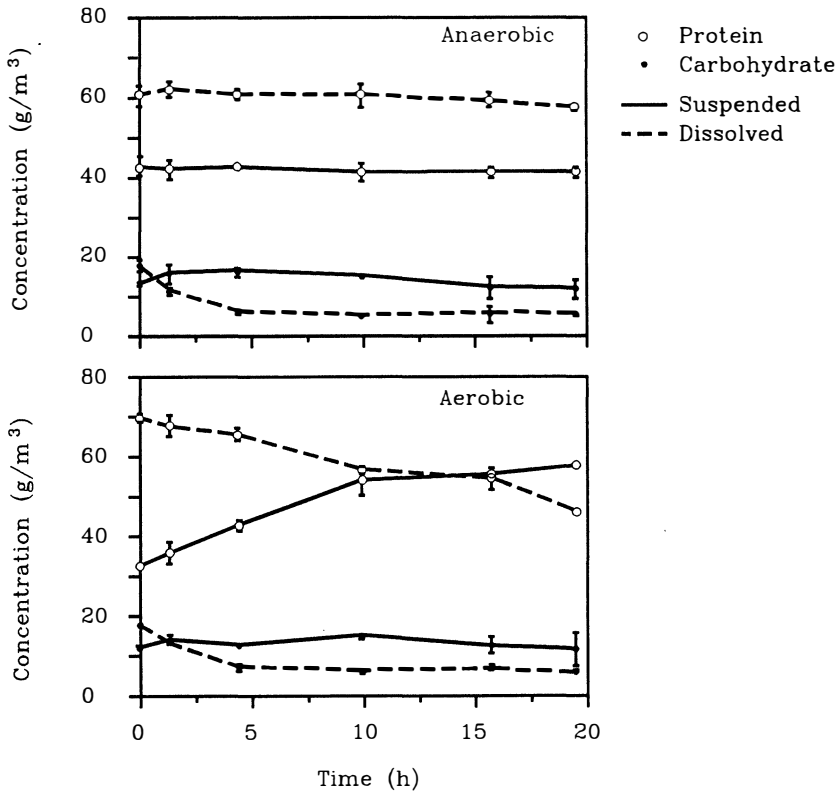

Figure 3. Changes in the composition of protein and carbohydrates in "young" wastewater at 20°C.

An example of changes in bulk water concentrations of proteins and carbohydrate (measured according to Lowry et al., 1951 and Gaudy, 1962) is shown in Figure 3 (Raunkjær, unpublished results). Young domestic wastewater with an age of about 2 hours and a total COD around 430 mg/l was analysed for changes in soluble and particulate fractions (filtered on 1.2 μm filter) during incubation at aerobic and anaerobic conditions. Soluble carbohydrate was, in contrast to particulate carbohydrate in both cases, rapidly removed from the wastewater until a constant level was found after 5 hours. This removable part is believed to be easily degradable starch and similar compounds while the remaining part could be slowly degradable cellulose. The presence of oxygen was important for the transformation of proteins. In aerobic systems the suspended proteins showed a significant increase during the experiment, indicating a growth of the bacterial biomass. Secondly, the soluble part was reduced significantly. In the absence of oxygen no significant changes in the protein pool were observed. These results are in accordance with other results from anaerobic systems (Breure et al., 1986, McInerney, 1988) showing that hydrolysis of starch takes place before significant hydrolysis of proteins. This experimental example demonstrates that both the organic component, the size fraction and the turnover rate in the wastewater were affected by the presence of oxygen. So far we are not able to quantify these transformations. Also growth and detachment of the biofilm is important for the organic constituents in the wastewater, and these processes are probably more important than the bulk water processes due to the high biomass and activity in the biofilm.

It is possible to make a rough estimate of the total consumption of organic matter on the basis of consumed electron acceptors. Table 4 shows some examples where the consumer organic matter/consumed electron acceptor was calculated in COD units for different electron acceptors. Acetate was in all cases electron donor. Also the produced biomass is shown. The stoichiometry was based on yield coefficients (g biomass produced/g organic matter consumed) of 0.45, 0.35 and 0.10 for growth with oxygen, nitrate or sulfate as electron acceptor respectively (Bouwer and Cobb, 1987, Widdel, 1988). Using these stoichiometric coefficients and the equations for oxygen, nitrate and sulfate consumption in either gravity sewers or pressure mains (equations shown in Tables 2 and 3), it is possible to make a rough estimate of the change in total organic matter when sewer characteristics and residence time is known. There are, however, several limitations. For instance no microbial decay and predation by protozoans or higher animals are included. Also a process such as fermentation is not taken into account although it is important in pressure mains. Furthermore, shortcomings of the rate equations shown in Table 2 are not yet solved as indicated later.

It is more difficult quantitatively to predict changes of the organic constituents in the wastewater. It may to some extent be estimated from the information give in Tables 1, 2, 3 and 4. In the presence of oxygen or nitrate, significant growth of bacteria occurs, and a part of the soluble protein disappears while the bacterial protein increases (Figure 3). In anaerobic systems the growth rate is usually low for fermenters and sulfate reducers, and they produce VFA, especially acetate, but it has not been quantified yet. Although it is possible to estimate or measure the sulfide build-up, it is presently not known when the sulfate reducers predominantly oxidize the organic matter to acetate or to CO_2. Both types of bacteria can be present in sewer systems (Nielsen and Hvitved-Jacobsen, 1988).

The presence of ammonia in wastewater is often in focus. No net ammonia is produced during growth in the presence of oxygen (and nitrate). However, hydrolysis and respiration of proteins produces ammonia and although a part is reassimilated for growth a net production may occur. Besides this, an unknown amount is produced from fermentation deep in the biofilm. In anaerobic systems ammonia is usually observed to appear during fermentation, but also here it is not quantified.

Table 3. Kinetic parameters from biofilters and rotating drum experiments, $r_a = k_{1/2a} C^{0.5} (g/m^2 \cdot h)$

Substrate	Type of reactor	temp. °C	half order surface rate $(k_{1/2a})$ $g^{1/2}/m^{1/2} \cdot h$	ref.
oxygen (O_2)	rotating drum (methanol, acetate or glucose)	10-13	0.14-0.18	Jansen and Harremoës 1985
nitrate (NO_3-N)	down-flow filter	18	0.03-0.05	Ref. in Harremoës et al., 1980
	rotating disc	20	0.13	Watanabe and Ishiguro, 1978
	rotating disc	20	0.084	Shieh, 1982
	rotating drum (methanol)	22	0.08-0.10 0.036-0.10	Harremoës et al., 1980 Jansen and Harremoës, 1985
Organics (as COD)	rotating drum (acetate, glucose)	11	0.13-0.14	Jansen and Harremoës, 1985
	rotating disc (glucose)	20	0.16	Onuma and Omura, 1982

Table 4. Stoichiometric relation between consumed organic matter (acetate), produced biomass and electron acceptor

Electron acceptor	g acetate consumed/ g e-acceptor consumed (gCOD/g e-acceptor)	g biomass produced/ g e-acceptor consumed (gCOD/g e-acceptor)
Oxygen (O_2)	2.50	1.50
Nitrate (NO_3-N)	4.84	1.74
Sulfate (SO_4-S)	2.24	0.24

EXPERIENCES FROM OTHER BIOFILM TYPES

The biofilm processes taking place in the sewer system are quantified using simple empirical models which do not take biofilm kinetics and dynamics into account. Other biofilm processes in wastewater treatment are significantly better understood and modelled. The sewer biofilm system differs from other biofilm systems in two important ways: very high organic loading is typical and high shear stress at the biofilm surface due to very high flow velocity in the bulk water may occur frequently. Experience from different kinds of biofilters and rotating drum reactors has shown that zero and half order kinetics usually describe the kinetics fairly well for consumption of oxygen, nitrate, sulfate and different organics (Harremoës et al., 1980, Jansen and Harremoës, 1985, Nielsen and Hvitved-Jacobsen, 1988). It may therefore be possible to extrapolate some of these results to work also in sewer pipes, and some examples of such equations are shown in Table 3.
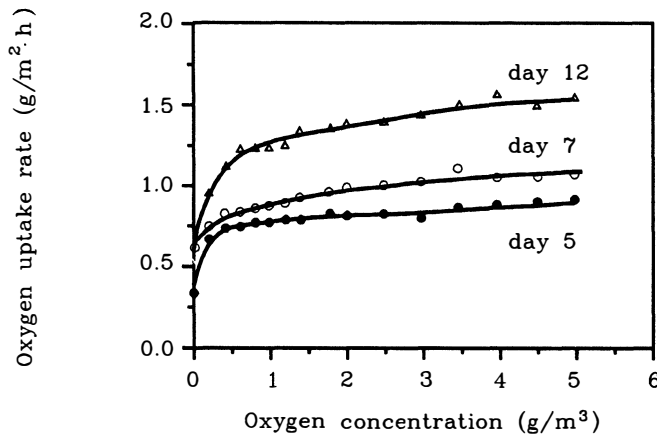
Figure 4. Kinetics of oxygen uptake in biofilms grown on domestic wastewater. Biofilm thickness was around 1000 μm.
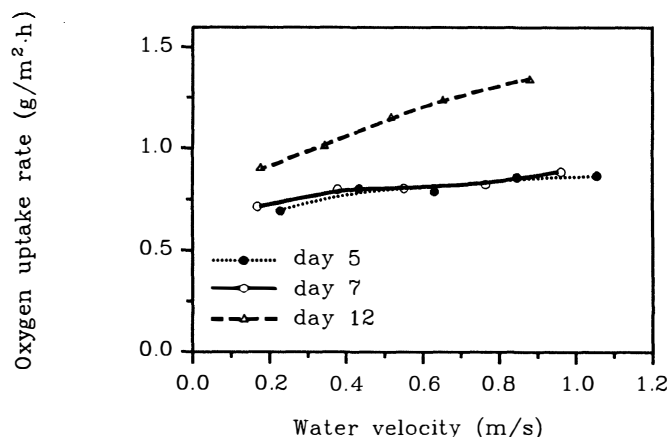
Figure 5. Effect of water flow velocity on oxygen consumption rate in biofilm grown on domestic wastewater. Oxygen concentration was 2 mg/l and biofilm thickness was around 1000 μm.

To evaluate some of these equations we have made some investigations on biofilms grown on domestic wastewater using a rotating drum reactor. Some results are shown in Figures 4 and 5 (Norsker, unpublished results). The biofilm was grown at 20°C with a very high organic loading, with a total COD of around 20 $g/m^2 \cdot h$. Hydraulic retention time was 22 minutes, peripheral flow velocity in the reactor was around 1 m/s (shear stress 2.0 - 2.6 N/m^2) and the oxygen concentration was kept at around 2 mg/l. Figure 4 shows an example of the oxygen uptake rate at varying oxygen concentrations. The uptake rate was found to be dependant on biofilm age and oxygen concentration. A high rate was found even at concentrations below 0.5 mgO_2/l. A slight concentration dependency was found at higher oxygen concentrations. The result does not agree very well with the half order kinetics predicted from equations shown in Table 3, or with the empirical equation (Table 2, no. 5) which predict a direct proportionality between rate and oxygen concentration. The reason could be found in the biofilm surface structure. It was very fluffy and rough, not smooth as assumed when using half order kinetics. Also Harremoës et al. (1980) described this phenomenon when the biofilm grew older. In real sewer systems the biofilms actually grow old and the high organic loading and the high shear stress probably create an extremely rough biofilm which significantly affects transportation and utilization of substrates.

Also the water velocity affected the oxygen consumption rate (Figure 5). The rates for two biofilm ages increased only slightly when the flow velocity increased from 0.2 to 1 m/s, while a higher dependency was found in a third case when the biofilm had grown older. Other investigators using a rotating drum reactor or a rotating disk usually have found a maximum constant oxygen uptake rate at velocities above 0.8-0.9 m/s. They then concluded that liquid film and external mass transfer resistance were insignificant above this velocity (La Motta, 1976, Shieh and Mulcahy, 1986). In contrast, the empirical equation from a sewer system (Table 2, no. 5) indicates that the flow velocity remains important also at high flow velocities in real sewer systems. Also Boon and Lister (1975) and Matos and de Sousa (1991) indicate that increased flow rates in real sewers cause higher oxygen uptake rates. However, due to different design of reactors and sewer pipes it is difficult to compare the flow velocities from the different studies and the effect on the biofilm surface. Instead information on shear stress ought to be included, but only a few measurements exist (e.g. Rittman, 1989, Characklis, 1989).

These experiments with biofilms grown on domestic wastewater at very high organic loading and shear stress showed that the usual biofilm kinetics do not work in an adequate way. The flow conditions and the surface structure should be taken into account, maybe by introducing an outer part of the biofilm with high turbulence (eddy diffusion) as proposed by Siegrist and Gujer (1985).

Little research has been conducted on the modelling of activities in multispecies biofilm. Some attempts have been done, e.g. by Gujer and Wanner (1990) and Chen et al. (1989), but so far such models have not been developed and calibrated to the conditions in sewer systems. Therefore, the above-mentioned empirical observations and equations still seem to be the best way to estimate the changes in wastewater composition during transportation.

CONCLUSIONS

The different chemical, physical and biological transformations of wastewater in sewer systems are very complex and can cause significant changes in the composition during transportation. Although the transformations cause many problems due to e.g. sulfide production and problems with optimizing the advanced wastewater treatment, the knowledge of the transformations is very limited and no advanced models of the processes in sewer systems exist today. However, there exist several simple empirical equations to estimate oxygen consumption rates, sulfide production rates and BOD removal rates in both gravity sewers and pressure mains. Using these equations significant changes in the wastewater composition during transportation can be calculated. Before these equations are used, it is very important to take into account the different traditions in design and operation of sewers, the origin and composition of wastewater and the climate. By evaluating simple as well as more complicated biofilm models it was found that two important characteristics for sewer systems, namely very high loading and high shear stress to the biofilm, create a multispecies biofilm with a very rough surface structure. In consequence existing biofilm models may not predict microbial changes well. Also the transformations of individual organic constituents in bulk water due to microbial activity was shown to be significant. More quantitative studies on organics are needed in full scale sewer pipes as well as laboratory scale focusing on bulk water versus biofilm reactions during aerobic and anaerobic conditions.

ACKNOWLEDGEMENTS

SYMBOLS

The following symbols are used in this paper:

C_{BOD}	biochemical oxygen demand (mgO$_2$/l)
C	substrate concentration (mg/l)
S_O	actual oxygen concentration (mg/l)
$S_{O,m}$	saturated oxygen concentration (mg/l)
C_{COD}	chemical oxygen demand (mg o$_2$/l)
S_{COD}	soluble COD (mg o$_2$/l)
d_m	hydraulic mean depth (m)
g	gravitational acceleration (m/s^2)
k	rate constant (-)
$k_{1/2a}$	half order rate constant (g$^{1/2}$/m$^{1/2}$·h)
$K_L a$	reaeration coefficient (1/h)
Q	flow (l/s)
r_a	biofilm surface rate, flux (g/m^2·h)
r_w	bulk water oxygen uptake rate (g/m^3·h)
r_s	sediment oxygen uptake rate (g/m^3·h)
r_b	biofilm oxygen uptake rate (g/m^3·h)
s	slope (m/m)
S_S	sulfide concentration (mg/l)
S_{SO4}	sulfate concentration (mgSO$_4$/l)
S_{SO4-S}	sulfate-sulphur concentration (mgS/l)
t	time (hours)
T	temperature (°C)
u	flow velocity (m/s)

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