Modeling the Formation and Fate of Odorous Substances in Collection Systems

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ABSTRACT: A conceptual model that simulates the formation and fate of odorous substances in branched collection systems is presented. The model predicts the activity of the relevant biomass phenotypes under aerobic, anoxic, and anaerobic conditions in force mains and gravity sewers. The formation and fate of individual, malodorous substances in the bulk water, biofilms, and sediments are modeled. The release of odorous compounds from the bulk water to the sewer gas phase, their fate in the gas phase, and their subsequent release into the urban atmosphere is simulated. Examples of model application include the prediction of hydrogen sulfide and malodorous fermentation products from force mains and gravity sewers. Water Environ. Res., 80, 118 (2008).

KEYWORDS: odor, hydrogen sulfide, modeling, fermentation, force main, gravity sewer.

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Introduction

Odors from collection systems are a well-known nuisance that regularly give rise to complaints (Stuetz and Frechen, 2001). The emission of malodors are often observed from large intercepting sewers with little slope or at locations downstream of force mains (Hvitved-Jacobsen, 2002). At the source, however, most wastewaters are not particularly odorous; for example, household wastewaters possess a more or less neutral hedonic tone, as do many industrial wastewaters.

Whether the hedonic tone becomes more unpleasant and/or the odor intensity increases during transport in the collection system depends on factors such as wastewater temperature, organic matter content, organic matter quality, oxidation–reduction (redox) conditions, and transport time. A crucial factor in the formation and fate of odorous substances in collection systems is the type of electron acceptors present and the type and quality of electron donors. In this respect, the central electron acceptors are oxygen, nitrate, organic matter, and sulfate, and the central electron donors are organic matter and sulfide (Stuetz and Frechen, 2001).

Generally, odorous substances are formed by degradation of organic matter in the absence of oxygen and nitrate (anaerobic conditions). When bulk water conditions are aerobic (i.e., when oxygen is present) or anoxic (i.e., when oxygen is absent and nitrate is present), organic matter is degraded without the formation of strong odors.

The substances potentially causing odors from collection systems are manifold. One important substance is hydrogen sulfide (H₂S), which, in addition to being malodorous, is toxic and the cause of concrete and metal corrosion. Other important malodorous substances possessing low odor thresholds are the products of fermentation and are organic in nature (i.e., indole, skatole, volatile fatty acids, amines, mercaptanes, organic sulfluides, aldehydes, and ketones). Hwang et al. (1995), one of the few studies concerned with measuring specific odorous substances in sewers, found high concentrations of the above-mentioned substances in the wastewater of sewers.

The fate of odorous substances formed—or their precursors—depends on the redox conditions and on the design of the collection system. Where redox conditions change from anaerobic to aerobic or anoxic, part of the odorous substances formed during anaerobic conditions are oxidized to nonodorous substances, while others remain undegraded (Hwang et al., 1995). Where the sewer is only flowing part-full, odorous substances are released through the free water surface and into the sewer gas phase, where they are either oxidized on the moist pipe walls or from where they can escape to the urban atmosphere (Hvitved-Jacobsen, 2002). Such conditions exist when the collection system contains long force mains or when the geometry of gravity sewers in combination with flow conditions, temperature, and wastewater quality causes anaerobic conditions.

It is the objective of this study to present a conceptual odor model for collection systems, integrating the latest understanding of sewer processes and the formation and fate of odorous substances. The model concept includes the relevant physical, chemical, and biological processes under aerobic, anoxic, and anaerobic conditions. The application of the model is exemplified by simulations of force mains and gravity sewers.

Methodology

Because the formation and fate of odorous substances predominantly are biological processes, the core of a model for simulation of formation and fate of odorous substances in collection systems must be the activity of the relevant biomass phenotypes. A model must also take substrates and relevant breakdown pathways into account and must simulate the major redox processes, including processes of a purely chemical and physicochemical nature. The processes take place in bulk water, biofilms, and sediments; therefore, each of these phases must be included, as must the relevant mass transfers across the phase boundaries. Owing to the fact that odorous substances are formed in the water phase but cause nuisances in the urban atmosphere, mass transfer between bulk water...
and sewer gas phase must be included in the concept, as must ventilation along the sewer line and the release of sewer gas into the urban atmosphere.

A model that takes into account the processes relevant for the simulation of electron acceptors and electron donors in collection systems has evolved over the last decades. The concept has been formulated in mathematical terms as a research tool under the name Wastewater Aerobic/Aerobic Transformation in Sewers (WATS), and comprises the work of numerous researchers (i.e., as summarized by Hvitved-Jacobsen [2002]). The parts developed first concerned aerobic and anaerobic transformations of organic matter and sulfur in bulk water, biofilms, and sediments. At present, it is attempted to integrate odor and corrosion to the WATS model; therefore, the model presented in this study makes use of preliminary and yet unpublished knowledge on nitrogen transformation, odor formation, and the fate of sulfide in the sewer gas phase.

**Odorous Substances.** Odors from sewers are caused by the combined effect of numerous volatile substances, both organic and inorganic in nature. The substances are present in highly varying concentrations, and many are difficult and time-consuming to measure, making it impractical to determine all relevant substances when assessing odor intensity and hedonic tone (Stuetz and Frechen, 2001). Furthermore, it has not been possible to relate the concentrations of individual odorous substances to the resulting odor impression.

When assessing odors by simulation of the formation and fate of chemical compounds, the need arises for indicator substances that are representative of the overall odor impression and can be readily measured and modeled. The potential odorous substances in wastewater of sewers are primarily sulfide and volatile fermentation products (volatile organic compounds [VOCs]). Sulfide especially has been suggested as an odor indicator, partly because there is often some correlation between the odor of raw wastewater and its sulfide content and partly because it is rather simple to measure, compared with most organic odorous substances (i.e., Thistlthwayte and Goleb, 1972).

However, using sulfide as the sole odor indicator is problematic. Even though sulfide is formed under conditions similar to other odorous substances (i.e., under anaerobic conditions and in the presence of readily biodegradable substrates), the breakdown pathways of sulfide differ significantly from those of volatile fermentation products. As not all fermentation products are also malodorous, it is, from a modeling point-of-view, relevant to use both sulfide and selected fermentation products as odor indicators.

**Aerobic Transformation of Organic Carbon and Sulfur.** A model for the aerobic carbon cycle in collection systems has been developed and experimentally validated under laboratory and field conditions (i.e., Almeida et al., 2000; Hvitved-Jacobsen et al., 1998; Vollertsen and Hvitved-Jacobsen, 1999; Vollertsen et al., 2005). The concept is illustrated by the right side of Figure 1 and shows how hydrolyzable substrates are transformed into readily degradable substrate and used for the growth and maintenance of heterotrophic biomass. Two and, in some cases, three fractions of hydrolyzable substrates are included, because experimental evidence has shown such a fractionation to be required (Vollertsen and Hvitved-Jacobsen, 2002). Biomass decay is not included in the concept, as this process is of minor importance under sewer conditions (Hvitved-Jacobsen et al., 1998).

Where conditions are aerobic, not only organic carbon acts as an electron donor, but also sulfide. The aerobic sulfur cycle is outlined on the left side of Figure 1 and has been studied in bulk water and biofilm by Nielsen et al. (2003, 2006) and Nielsen, Hvitved-Jacobsen, and Vollertsen (2005). They quantified the in-sewer sulfide oxidation and found oxidation to take place in both the bulk water and the biofilm. Furthermore, they report that biological sulfide oxidation and chemical sulfide oxidation, in general, are of comparable magnitude.

**Anaerobic Transformation of Organic Carbon and Sulfur.** The anaerobic sulfide formation process has been studied by numerous investigators, and Pomeroy et al. (1985) have quantified this process (Figure 2). The anaerobic carbon transformation processes, in terms of anaerobic hydrolysis and fermentation, were studied by Tanaka, Hvitved-Jacobsen, and Horie (2000) and integrated with the sulfide formation process. Tanaka, Hvitved-Jacobsen, Ochi, and Sato (2000) validated this part of the concept through laboratory and field experiments.

The processes that cause formation of specific organic odorous substances and their fate in collection systems have previously received little or no attention. Consequently, little is known of the complex microbial formation and breakdown pathways and the corresponding kinetics of these substances (Stuetz and Frechen, 2001). The level of detail applied when modeling the formation

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**Figure 1—Main processes for transformations of organic carbon and sulfur under aerobic conditions in a gravity sewer.**

- **Sulfide oxidation and corrosion.**
- **VOC's.**
- **Odors.**
- **Ventilation.**
- **Gaseous sulfide.**
- **Metal sulfides.**
- **Sulfate, elemental sulfur, thiosulfate.**
- **Dissolved oxygen.**
- **H₂S + HS⁻.**
- **Carbon cycle.**
- **Sulfur cycle.**
- **Wastewater and biofilm.**
- **Gas phase.**

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and fate of organic odorous substances must reflect the state of knowledge and must consequently be kept correspondingly simple. In the concept presented, the organic malodorous substances are lumped and modeled as a subset of the total amount of fermentation products; for example, the formation and breakdown kinetics of malodorous substances is assumed identical to those of fermentation products in general.

Anoxic Transformation of Organic Carbon and Sulfur. Anoxic conditions seldom prevail in sewer systems. However, when controlling odor and corrosion, especially in force mains, the addition of nitrate has proven an efficient approach (Aeso, et al., 1997). Nitrate prevents the presence of sulfide and fermentation products—a fact that is well-known from other environmental systems, such as wastewater treatment plants (Figure 3).

The oxidation of organic matter under anoxic conditions in collection systems is not yet fully understood; however, laboratory studies on the bulk water processes indicate that nitrite accumulates as an intermediate (Abdul-Talib et al., 2002). Unpublished laboratory-scale investigations show that nitrate reduction in sewer biofilms also gives rise to nitrite accumulation (Vollertsen et al., in prep). High nitrate concentrations will inhibit the biological processes, as nitrite is in equilibrium with the toxic compound nitrous acid (HNO₂) (Weon et al., 2002). However, in real sewer systems, nitrite accumulation has so far not been reported; therefore, the practical importance of nitrite as an intermediate is questionable, and nitrite accumulation has consequently been excluded from the model.

Yang et al. (2005) reports anoxic biological sulfide oxidation to take place in the wastewater of sewers, at a significantly lower rate than aerobic biological sulfide oxidation. However, they did not observe chemical sulfide oxidation by nitrate or nitrite.

Precipitation of Sulfide. Depending on pH and redox conditions, sulfide is precipitated by heavy metals. In the wastewater and biofilms of collection systems, iron and zinc are especially significant, with respect to sulfide precipitation (Nielsen, Lens, Hvitved-Jacobsen, and Vollertsen, 2005). In many wastewaters, the naturally occurring precipitation capacity is approximately 0.5 g S m⁻³; however, the empirical knowledge on precipitation capacities of different wastewaters is still rather limited (Hvitved-Jacobsen, 2002).

Mass Transfer Over the Air–Water Interface. The concept for mass transfer is established based on the physicochemical characteristics and conditions at the air–water interface. Where a free water surface exists, volatile substances are transferred over the...
Table 1—The mathematical formulation of the sulfur segment of the model concept depicted in Figures 1 to 3.*

<table>
<thead>
<tr>
<th>Process</th>
<th>( \frac{d}{dt} S)</th>
<th>( \frac{d}{dt} S_{\text{O}_{2}} )</th>
<th>( \frac{d}{dt} S_{\text{S}} )</th>
<th>( \frac{d}{dt} S_{\text{H}_{2}S} )</th>
<th>( \frac{d}{dt} S_{\text{S}<em>{2}O</em>{4}} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Formation of ( \text{H}_{2}\text{S} ) in biofilm</td>
<td>+1/( r_{1} )</td>
<td>-1/( r_{1} )</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chemical oxidation of sulfide in bulk water</td>
<td>-1/( R_{\text{CH}_{2}S} )</td>
<td>-1/( r_{2} )</td>
<td>+1/( r_{2} )</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biological oxidation of sulfide in bulk water</td>
<td>-1/( R_{\text{BH}_{2}S} )</td>
<td>-1/( r_{3} )</td>
<td>+1/( r_{3} )</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biological oxidation of sulfide in biofilm</td>
<td>-1/( R_{\text{BH}_{2}S} )</td>
<td>-1/( r_{4} )</td>
<td>+1/( r_{4} )</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Emission of ( \text{H}_{2}\text{S} ) from water phase to gas phase</td>
<td>-1/( R_{\text{BH}_{2}S} )</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{H}_{2}\text{S} ) oxidized to sulfuric acid on concrete surface</td>
<td>-1/6</td>
<td>3/2</td>
<td>3/2</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* \( r_{i} \) = process rate (days\(^{-1}\)); \( S_{O_{2}} \) = dissolved oxygen concentration (g O\(_{2}\) m\(^{-3}\)); \( S_{2O_{2}} \) = sum of \( \text{H}_{2}\text{S}, \text{HS}^{-}, \text{and S}^{2-} \) concentrations in the water phase (g S m\(^{-3}\)); \( S_{\text{S}_{2}O_{4}} \) = sulfate concentration (g S m\(^{-3}\)); \( P_{\text{H}_{2}S} \) = \( \text{H}_{2}\text{S} \) partial pressure in the gas phase (ppm); \( S_{\text{S}_{2}O_{4}} \) = sulfuric acid concentration in the water film of moist sewer surfaces (g S m\(^{-3}\)); \( R_{\text{CH}_{2}S} \) = stoichiometric constants for sulfide oxidation (−); \( R_{\text{BH}_{2}S} \) = universal gas constant (J mol\(^{-1}\) K\(^{-1}\)); \( T_{x} \) = temperature (K); \( V_{g} \) = gas volume (m\(^3\)); and \( V_{w} \) = water volume (m\(^3\)).

**The Mathematical Model.** The concept depicted in Figures 1 to 3 can be formulated in terms of differential equations for each model component, accounting for the biomass and substrate dependencies of the individual processes. For the purpose of illustrating the mathematics of the concept, a segment of the formulated equations is shown in.

The notation applied is similar to what is used by several environmental models (i.e., activated sludge models). The matrix must be understood as the differentials of the first row being equal to the sum of the processes listed in the respective columns. As an example, the change in hydrogen sulfide partial pressure in the sewer gas phase (\( P_{\text{H}_{2}S} \)) is the sum of the emission of hydrogen sulfide from the water phase and the hydrogen sulfide gas oxidized on the moist surfaces of the sewer gas phase (eq 1 terms are identified in Table 1).

\[
\frac{dP_{\text{H}_{2}S}}{dt} = \frac{R_{T}V_{w}}{32} V_{w} T_{S} - r_{6} \tag{1}
\]

The complete model contains 15 coupled differential equations that are solved by a finite difference method together with a simple

air–water boundary. Mass is transferred in both directions, depending on the bulk water being supersaturated or unsaturated, with respect to the sewer gas phase. The mass-transfer kinetics of a substance is identical in either direction, and the transfer rate depends solely on the magnitude of the concentration gradient.

The resistance to mass transfer over the air–water boundary lies both in the water phase and in the gas phase. However, for substances having a high Henry’s constant (i.e., being volatile in dilute aquatic solutions), the resistance to mass transfer is primarily in the water phase, and the transfer rate is determined by physical conditions of the water phase alone. Such is the case for oxygen, hydrogen sulfide, ethyl mercaptan, and methyl mercaptan. For malodorous substances with a low Henry’s constant, the resistance to mass transfer lies in the gas phase alone, and the mass transfer depends on physical conditions in the gas phase and not on, for example, turbulences in the water phase. Such substances are ammonia, volatile fatty acids, and methylamine. Another large group of malodorous substances are found between these boundaries, and the mass transfer of these substances depends on physical conditions in both phases (Hvitved-Jacobsen, 2002; Liss and Slater, 1974).

For two substances with similar mass transfer properties, the ratio of their respective mass transfer rate constants is also a constant and independent of physical characteristics, such as turbulence (Liss and Slater, 1974). The ratio is independent on temperature if the Henry constants of the two substances have similar temperature dependencies. These theoretical facts have been verified experimentally by Yongsiri, Vollertsen, and Hvitved-Jacobsen (2004) and Yongsiri, Vollertsen, Rasmussen, and Hvitved-Jacobsen (2004) and can be applied in the estimation of the mass transfer rate constant of a substance for which the mass transfer properties are not experimentally determined.

Oxygen mass transfer from the gas phase and into the bulk water (reaeration) has been intensively studied, and empirical equations describing reaeration are well-established for many aquatic systems (i.e., Jensen, 1995). Oxygen has a high Henry’s constant and mass transfer characteristics similar to that of many volatile, odorous substances. Together with the ratio between the mass transfer rate constant of oxygen and that of the substance in question, the empirical reaeration equations are applied to simulate the mass transfer of odorous substances and other VOCs with high Henry’s constants (WEF, 1998). The transfer of substances for which the main transfer resistance is in the gas phase (i.e., substances having a low Henry’s constant) is not simulated because of a lack of solid empirical knowledge on the mass transfer of such substances.

**Ventilation and Sewer Gas Release.** The microbial and chemical processes in wastewater and biofilms of collection systems occur on a timescale that makes it irrelevant to simulate nonuniform or unsteady flow conditions. The hydraulics applied in the simulations presented are consequently modeled as steady and uniform flow applying the Colebrook-White equation. The composition of the wastewater entering the sewer is assumed quasi-steady. Therefore, the dispersion of substances is not considered, and plug flow is assumed when simulating wastewater transformations. Furthermore, as odor nuisances primarily are dry-weather phenomena, only dry weather conditions are addressed.

Similar to the flow of water, gas is conveyed along the sewer line. However, where it is the gravitational force that causes the water to flow, it is mainly the drag force of the flowing water acting on the gas volume that causes ventilation. The ventilation in sewers has received some attention; Madsen et al. (2006) report natural sewer gas movement to occur at 10 to 50% of the wastewater flow velocity.

The release of sewer gas out of the collection system and into the urban atmosphere is not well-understood, but its magnitude is expected to be governed by sewer geometry, slope, water level variations, water velocity, and size, number, and location of openings to the atmosphere (Bell et al., 1995; Corsi et al., 1992; Madsen et al., 2006; Olson et al., 1997; Pescod and Price, 1982). In the simulations presented, sewer gas release is simulated through openings in manhole covers only. Attempts have been made to quantify sewer gas release, and parameters such as windspeed, differences in atmospheric pressure, and differences in temperature have been adjusted to govern the magnitude of gas release (Olson et al., 1997).
Euler approach. A more complex and more accurate solution scheme is not an option, as the differential equations are nonlinear, rather complex, and coupled. The consequence of this rather crude solution scheme is a short time-step of approximately 1 to 5 seconds per step.

Details on the model equations and experimentally determined model parameters are found in Hvittved-Jacobsen (2002), Nielsen et al. (2006), Tanaka, Hvittved-Jacobsen, and Horie (2000), Vollertsen and Hvittved-Jacobsen (1999), and Vollertsen et al. (2005).

Results and Discussion

Because of the many interlinked processes that proceed in the different phases of a collection system, the outcome of any simulation becomes strongly dependent on not only the physical characteristics of the system, but also on numerous factors, such as flow characteristics, wastewater composition, wastewater pH, temperature, airtightness of the system, and pipe materials. The possible configurations and operational conditions are consequently many; therefore, this study is restricted to simulations that exemplify the application of the model concept.

In the following, sulfide and methyl mercaptan are chosen as odor indicators. The two substances are chosen because they are known to contribute significantly to the overall odor intensity (Hvittved-Jacobsen, 2002; Hwang et al., 1995). Methyl mercaptan is chosen as an example of a volatile fermentation product with a low odor threshold value. However, the formation and breakdown of methyl mercaptan in sewers is not well-understood. In this study, the degradation of methyl mercaptan in the bulk water is assumed to occur at the same rate as the bulk of fermentation products formed under anaerobic conditions (Figure 2). In the gas phase, methyl mercaptan might be reduced, to some degree, by autotrophic, sulfide-oxidizing biomass. However, the extent of this process is not known, and, in this study, methyl mercaptan is assumed inert in the sewer gas phase.

Example 1—Formation of Odorous Substances in Force Mains. Force mains are important sources for odorous substances, because the wastewater typically is anaerobic for several hours. The substances are formed both in the biofilm and in the bulk water; however, the biofilm is the main contributor of hydrogen sulfide, as the growth rate of sulfate-reducing bacteria is low, and a long biomass residence time consequently is a prerequisite. Figure 4 is an example of the formation of sulfide and methyl mercaptan in a force main 400 mm in diameter, with an average flowrate of 0.025 m$^3$ s$^{-1}$ and a chemical oxygen demand (COD) content of 670 g COD m$^{-3}$. The formation of methyl mercaptan is not well-understood, and the formation is simply assumed to be 0.5% of the fermentation products formed.

Significant concentrations of both sulfide and fermentation products (i.e., methyl mercaptan) are formed during the 5 hours and 5 minutes of residence time in the force main (Figure 4). These substances are likely to cause a nuisance in a downstream gravity sewer. One way of controlling that problem is to add nitrate at the pumping station. Figure 5 depicts a case where 20 g nitrate-nitrogen (NO$_3$-N) m$^{-3}$ is added. The nitrate is completely reduced after 3.2 km, upon which the force main becomes anaerobic some 0.8 km upstream of the discharge manhole, resulting in a slight formation of sulfide and other odorous substances.

As exemplified by Figure 5, the addition of nitrate is an efficient tool to avoid the formation of odorous substances. Furthermore, the handling of nitrates is rather unproblematic. On the other hand, nitrates are typically rather costly, often limiting the application of this strategy to situations where other and cheaper strategies are insufficient.

Example 2—Formation of Odorous Substances in Gravity Sewers. The formation of odors in gravity sewers depends on the balance between reaeration and oxygen-consuming processes. The oxygen consumption rate increases with increasing concentrations of dissolved oxygen, whereas the reaeration rate increases with decreasing dissolved oxygen concentration (i.e., the two processes proceed in opposite direction, and a balance is obtained at some dissolved oxygen concentration). Sewers with low slope, large water depths, high temperatures, and concentrated wastewaters tend to obtain that balance at a rather low dissolved oxygen concentration, and they hereby become quasi-anaerobic. Under such conditions, odors are formed in gravity sewers; however, in general, they are formed at a lower rate compared with force mains, because the continuous supply of oxygen from the sewer gas phase limits the extent of the anaerobic processes.

An example of the release of methyl mercaptan and the corresponding dissolved oxygen concentrations in a gravity sewer with slopes varying from 0.05% to 0.5% is given in Figure 6. The pipe is constructed of concrete and has a diameter of 1 m. The flowrate is 0.25 m$^3$ s$^{-1}$, corresponding to water depths ranging from 0.55 to 0.30 m for the different slopes.

The simulations presented in Figure 6 show the change from quasi-anaerobic conditions at low slopes to aerobic conditions at high slopes. When the dissolved oxygen concentration is low, anaerobic processes take over, here illustrated by the formation of methyl mercaptan.
Example 3—The Fate of Odorous Substances. Figure 7 exemplifies the fate of sulfide and methyl mercaptan in a gravity sewer constructed of concrete. The gravity sewer receives 0.025 m$^3$ s$^{-1}$ of anaerobic wastewater from a force main, containing 0.15 g m$^{-3}$ methyl mercaptan and 3 g m$^{-3}$ sulfide at a pH of 7.5 (Hvitved-Jacobsen, 2002; Hwang et al., 1995). The gravity sewer has a diameter of 1 m, and the slope is 0.5% for the simulations shown in the upper graphs of Figure 7 and 0.05% for the simulations shown in the lower graphs. The gas flow velocity is 10% of the water flow velocity, and no gas is lost because of ventilation (i.e., all manhole covers are assumed hermetically sealed).

For the gravity sewer with a slope of 0.5% (Figure 7, upper graphs), the simulations show that both sulfide and methyl mercaptan are removed from the bulk water at approximately the same rate. At a slope of 0.05%, however, methyl mercaptan is removed significantly slower than sulfide (Figure 7, lower graphs). The main reason for these differences is twofold—(1) the release rate of both sulfide and methyl mercaptan is highest in the sewer with high slope, and (2) the oxygen concentration in the sewer with high slope is approximately 2 g m$^{-3}$, whereas the oxygen concentration in the sewer with low slope is close to 0 g m$^{-3}$ (i.e., in the first case the water phase is aerobic, while, in the second case, it is quasi-anaerobic). The reason that sulfide is low in the gas phase of the sewer with a slope of 0.05%, while methyl mercaptan is high, is that sulfide becomes oxidized on the moist pipe walls, while methyl mercaptan in this study is assumed inert in the gas phase.

These examples illustrate why the sulfide content of the gas phase or the bulk water is not always a good odor indicator; not only is the formation of the different odorous substances brought about by different processes and by different phenotypes of microorganisms, but also the release and breakdown patterns of different malodorous substances differ significantly.

Pipe Characteristics. The simulations shown in Figure 7 illustrate the complex dynamic relationship between the different...
processes of formation, removal, and mass transfer that governs some of the substances causing odors from sewers. Even though the different processes and their mutual interactions are complex, some simple relationships can be extracted by applying stochastic modeling. Figure 8 illustrates the importance of pipe characteristics on the occurrence of odorous substances along a gravity sewer line. The wastewater contains initially 0.15 g m\(^{-3}\) methyl mercaptan and 3 g m\(^{-3}\) sulfide at a pH of 7.5 and enters the gravity sewer through a sewer drop. A total of 500 simulations are performed, with randomly drawn combinations of pipe diameter, slope, and flow, keeping all other model parameters fixed (Figure 8). The parameters are drawn from linear distributions in the following intervals: 0.2 m ≤ pipe diameter ≤ 1.5 m, 0.01% ≤ slope ≤ 1%, and 0.1% ≤ percent of full flow ≤ 50%. The gas flow velocity is 10% of the water flow velocity, and no gas is lost because of ventilation (i.e., all manhole covers are assumed hermetically sealed).

The simulations illustrate how the ratio between hydrogen sulfide and methyl mercaptan differs along the sewer line (Figure 8). The propagation of both gases is affected by the water velocity, and high velocities tend to cause the gases propagate furthest. In the upstream sewer section, hydrogen sulfide is the dominant compound, whereas its role diminishes further downstream. The decrease in methyl mercaptan, on the other hand, is less pronounced.

Causes for the different behaviors of the two gases include a higher Henry’s constant for hydrogen sulfide than for methyl mercaptan, that sulfide is oxidized in the water phase under aerobic conditions, and that hydrogen sulfide is continuously oxidized on the moist concrete sewer walls. The different propagation pattern of hydrogen sulfide gas and methyl mercaptan gas in the sewer atmosphere illustrates that the odor composition changes along the sewer line and that hydrogen sulfide is not always a good indicator for the occurrence of odor. A similar conclusion was drawn by Stuetz et al. (1999), who found that hydrogen sulfide in the offgas from wastewater treatment plants did not correlate with the odor intensity.

Sewer Gas Release. A collection system is, in reality, never completely airtight, and sewer gas release will occur in any such system. Figure 9 shows the simulation of sulfide and methyl mercaptan in the gravity sewer with a 0.5% slope, which was also modeled in Figure 7. However, in this case, the sewer is not airtight, and 3.4 \(10^{-4}\) m\(^3\) m\(^{-1}\) s\(^{-1}\) is released to the urban atmosphere from manholes located every 50 m. The applied release rate is deduced from field experiments reported by WEF (1998).

Comparing Figure 7 with Figure 9, it is seen that gas release reduces the occurrence of methyl mercaptan in the gas phase, whereas the gas phase concentration of hydrogen sulfide is reduced to a lesser degree. The reason for this difference is that, while gas release is the only sink for gaseous methyl mercaptan, both gas release and oxidation on the concrete sewer walls are sinks for gaseous hydrogen sulfide. The comparison illustrates that a rather airtight gravity sewer system that receives anaerobic wastewater will cause higher odor intensities in case of a breach compared with a more open system. However, natural gas release alone is insufficient to completely control odors from the gravity sewer.
Odor Control. A possible approach for odor control of wastewater from a force main is to strip off volatile odorous substances into the gas phase, from where it is removed by forced ventilation. In the simulation example shown in Figure 10, the wastewater drops 2 m from a force main and into a gravity sewer. A simplified ventilation concept is applied, in which the forced ventilation removes all obnoxious gas from the first 1000 m of the gravity sewer, but does not affect the ventilation further downstream. Comparing Figure 10 with Figure 7 shows that forced ventilation, in combination with treatment of the emitted sewer gas, can be an effective method for controlling the gas phase concentrations of hydrogen sulfide and methyl mercaptan.

Another effective odor control method is to avoid anaerobic conditions (Figures 5 and 6). In the case of gravity sewers, this is best done in the design phase, where simulations of the redox conditions allow assessment of the risk of anaerobic conditions and the consequent odor effects. In the case of force mains, conditions can be kept aerobic by addition of air or pure oxygen or conditions can be kept anoxic by the addition of nitrate salts, with the latter solution being very effective. The model applied in this study is designed to simulate such scenarios also.

Conclusion

Malodors in collection systems are caused by a combination of numerous volatile substances with low odor threshold values, the formation and fate of which are complex and interlinked. However, current knowledge on in-sewer processes does allow simulation of substances with known chemical and biological characteristics. Hydrogen sulfide is a wastewater constituent that must be included in odor simulations; however, fermentation products also must be taken into account, because some of these products contribute significantly to the overall odor impression. Furthermore, even though the formation of fermentation products occurs under conditions similar to the formation of sulfide, the fate of fermentation products differ significantly from the fate of hydrogen sulfide.

In-sewer process modeling of the relevant electron acceptors, electron donors, and involved biomass phenotypes allows prediction of the formation, reduction, release, and fate of malodorous substances already in the design phase of a collection system. Thus, the means needed for optimizing the design of a collection system, with respect to odor control and possible odor hotspots, become available.

A reliable prediction of malodors from sewers calls for further and thorough studies of the involved biological, chemical, and physical processes. In depth knowledge on the formation, degradation, and release of the relevant compounds and thorough studies on liquid–gas mass transfer, ventilation, and gas release processes has the potential of turning conceptual odor modeling into an important tool for odor management. Odor simulation can be applied in the design of management strategies and makes it possible to assess the effect of a strategy before its implementation. Also, the consequences of changes in a catchment (i.e., in terms of establishment of new industries or the connection of new subcatchments) can be quantified by this methodology. In general, conceptual modeling of odors is recommended as a versatile tool for the design and management of collection systems.

The deadline to submit Discussions of this paper is May 15, 2008.

References


