Dissolved oxygen determination in sewers using flow hydraulic parameters as part of a physical simulation model

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ABSTRACT

This paper aims to develop a model for calculating the hydraulic and water quality parameters of wastewater within sewers. Information from the wastewater collection network and the transmission line in Birjand were used to verify the model performance. The parameters used for modelling quality changes include the yield constant for biomass ($Y_H$), the maximum specific growth rate ($\mu_H$), the saturation constant for dissolved oxygen ($K_{O2}$) and the saturation constant for readily biodegradable substrate within a biofilm ($K_{Sf}$), as well as the Gauckler–Manning–Strickler coefficient ($n$). They were selected from references and were verified at the calibration stage comparing measurements with the modelling values. Inputs of the created model are the average concentrations of dissolved oxygen and chemical oxygen demand of the incoming wastewater, the flow rate of wastewater at the exit point of the network, physical characteristics of the pipes and the height of drops within the sewer network. The amount of dissolved oxygen at different positions of the sewer network was calculated. The acceptable calculated sum of squares of errors and the correlation coefficient ($R^2$) of the calibrated model for dissolved oxygen were 1.6872 and 0.77, respectively.

Key words: COD changes within sewer, dissolved oxygen within sewer, physical simulation modelling, quality model calibration, wastewater network

HIGHLIGHTS

- A model was developed to calculate the hydraulic and qualitative parameters in sewers.
- The model was calibrated using the flow rate, organic strength and the dissolved oxygen.
- A sensitivity analysis of the dissolved oxygen change was performed.
- Dissolved oxygen was determined using the flow rate at the transmission line.
- The coefficient of flow change over time can be used to calculate dissolved oxygen.

INTRODUCTION

Sewer networks are urban infrastructure elements that collect wastewater and, in some cases, surface waters from residential and industrial areas and transfer them to wastewater treatment plants and/or to natural acceptors. Because wastewater contains a variety of biological and chemical substances, wastewater collection systems act as biological and chemical reactors, whereas wastewater treatment is typically considered only in the treatment plant, and biological processes within sewers are not commonly considered (Almeida 1999).

In addition to wastewater transport, one of the purposes of wastewater networks is to reduce the wastewater organic load. This service can be defined as a quality performance index (Heydarzadeh et al. 2019). On the contrary, the concentration of dissolved oxygen in the sewers is a major indicator of wastewater quality. The likelihood of reaching anaerobic conditions...
within sewers cannot be easily determined. At present, the measurement of dissolved oxygen in sewers is mostly done by installing relevant sensors in different positions of the network, which is expensive and presents operational problems.

Almeida et al. (2000) conducted a study on a sewer pipeline with a length of 7.2 km, a diameter of 1,000 mm, an average slope of 0.7% and the presence of a number of waterfalls. They found that the concentration of dissolved oxygen within sewers was always >1 mg/l. Their findings highlighted the importance of calculating the oxygen balance within sewers and identified the design conditions for maximizing the concentration of dissolved oxygen. The wastewater composition was found to be dependent on retention times. Long retention times, especially in aerobic conditions, led to a reduction of the concentration of biodegradable materials.

Chen & Leung (2000) conducted a study on oxygen consumption within a 1.5-km long municipal wastewater gravity pipeline. They measured the wastewater flow rate, the inlet and outlet oxygen concentrations, the specific oxygen uptake rate and adenosine triphosphate in the liquid phase and in the sediments. They observed that the difference in dissolved oxygen concentration between the inlet and outlet sections of the studied sewer pipe was approximately constant and equal to 3.1 mg/l, meaning that the oxygen consumption during wastewater transfer was constant and that the sediments are the main consumer of oxygen within sewers. The wastewater liquid-phase oxygen uptake rate varied around an average value of 17.7 mgO2/gSS/h over 24 h, while the oxygen uptake rate of the sediment phase decreased along the length of the pipe. They also reported that the mean levels of adenosine triphosphate in the liquid and sediment phases were almost constant, meaning that a large proportion of bacteria was present in both phases. Based on the results of their work, changes in suspended solids and total organic carbon in the liquid phase of wastewater are related to changes in the flow rate and indicate that higher flows are able to transfer larger amounts of organic matter and suspended solids than lower flows. The researchers measured changes in dissolved oxygen in the sewers, but did not provide a computational model for determining it.

Huisman et al. (2004) conducted a study on a main sewer line for identifying the main components acting on the oxygen balance in the sewers. By measuring the concentration of oxygen in the liquid phase and in the air above the wastewater along the pipe, they observed that the changes in the gas-phase oxygen were small and the concentration of dissolved oxygen in the liquid phase decreased along the sewers. The amount of oxygen uptake rate increases with increasing wet surface and dissolved oxygen, which indicates a greater effect of the biofilm layer and suspended biological matter on the oxygen uptake and the removal of organic matter. They also showed that the removal rate of soluble chemical oxygen demand (COD) in the sewer system could reach a considerable value of 30%. These researchers have carefully studied the effect of biofilm cellular respiration on the oxygen balance and COD removal within sewers, but have paid less attention to the role of suspended biomass, which is very effective in biological processes within sewers.

Huisman & Gujer (2002) conducted a study to investigate the re-aeration of wastewater within sewers. They examined several available equations and then calibrated one equation using information from different sources as well as information from four different sewer pipes. Then, the changes in re-aeration of the sewer system as a function of hydrodynamic conditions were investigated. They performed their experiments using a gas tracing-based method using sulfur hexafluoride and presented an equation to calculate the rate of oxygen transfer to wastewater. They calculated the actual hydrodynamic effect of the flow and slope on the gas-liquid-phase mass transfer in a hypothetical sewer, and they showed that the effect of the slope on the re-aeration parameter is much more important than the flow.

Ganigue & Yuan (2014) assessed the impact of oxygen injection on the production of methane and nitrous oxide in pressurized sewers. Oxygen injection was used as a way to control sulfide production. They argued that methane-producing microorganisms present in the inner layers of the biofilm resisted the inactivation due to the inability of oxygen to penetrate the inner layers, and therefore, methane production did not stop completely. Oxygen injection can also lead to the production of nitrous oxide by increasing the activity of nitrate-producing bacteria. They also concluded that despite a significant reduction in nitrogen, nitrous oxide production is negligible due to oxygen injection, and that nitrogen depletion is largely due to nitrification and denitrification processes occurring simultaneously. The researchers did not study the production of methane and nitrous oxide in gravity sewers that are affected by oxygen in the air above the sewage. Performing nitrification and denitrification processes in sewers can help to treat wastewater in the treatment plant. Therefore, the provision of a solution to increase these processes could be useful.

Ahrari (2015) investigated wastewater qualitative changes in the wastewater collection network of Birjand. She sampled different parts of the sewer network and measured various water quality parameters. Then, using geostatistical methods, she extracted selected variograms based on the relevant statistical indicators for each of the parameters in different sampling periods and determined the concentration distribution of the measured qualitative parameters for the Birjand sewer network.
using the Bayesian kriging method. She concluded that the concentrations of the two parameters, biochemical oxygen demand and COD, are higher in the northern parts of the city, which are further from the transmission line. Also, with the passage of wastewater through the network and approaching the transmission line, the concentration of these parameters decreases when wastewater approaches the transmission line, which indicates that wastewater treatment increases with the time wastewater is retained in the network. She merely measured the quality parameters of the wastewater at different points in the wastewater network and did not suggest a model to address the quality changes or calculate the amount of changes in the network according to the retention time of the wastewater.

Ebrahimi Raviz & Amini Rad (2016) examined the effect of biofilm formation and suspended biological mass in wastewater networks in decreasing solids, biochemical oxygen demand, COD, nitrogen and dissolved oxygen by constructing a wastewater collection network pilot in the laboratory. They also investigated the biofilm characteristics during the experiment. The biofilm formed on the inner surface of the tube was not uniform, and its thickness varied from 3 to 4.7 mm. Its surface density varied from 22.3 to 53.1 g of biomass/m². They reported the oxygen consumption rate at approximately 0.21 mg/l/min, the suspended solid concentration decrease rate at 82% and the maximum network removal efficiency at different loads for biochemical oxygen demand and COD at 59 and 54%, respectively. They observed that the changes in the concentration of nitrogen compounds were negligible and concluded that the pilot test did not lead to the growth conditions for nitrate-producing bacteria or for the nitrification process. Experiments were performed under pilot-controlled conditions with uniform flow, and the effect of high-flow velocities and leaching of the biofilm layer that occurs within sewers were not investigated. Also, the removal efficiencies reported by them are higher than the values reported by other researchers.

Xu et al. (2016) investigated the formation, structure, dissolved oxygen transfer and biofilm activity under various hydraulic conditions using microelectrode technologies as well as oxygen uptake rate. Their findings showed that the porosity of the biofilm decreases with increasing shear stress and tends to become a constant value at the end of the biofilm production. According to their findings, when the shear stress of the wall is 1.12, 1.29 and 1.45 Pa, the porosity of the biofilm in stable conditions is 69.1, 64.4 and 55.1%. The corresponding oxygen uptake rate values are 0.033, 0.027 and 0.022 mg/l/s, and the COD removal efficiency of sewers reaches 40, 35 and 32%, correspondingly. One of the effects of higher flow velocity in sewers is the increase of sewage re-aeration from the flow surface, which leads to increased removal of organic matter. Therefore, it seems necessary to determine the range of velocity that does not cause the biofilm layer to disappear and does not reduce the amount of wastewater re-aeration, which the researchers did not address in this study.

Najafzadeh et al. (2016) proposed two different artificial intelligence approaches such as Model Tree (MT) and the Evolutionary Polynominial Regression (EPR) to investigate sediment transport in sewer pipes using four dimensional parameters including volumetric concentration, total friction factor, non-dimensional particle size and the ratio of the hydraulic depth of flow to pipe diameter. They reported that MT and EPR models are comparable from the accuracy point of view, outperforming the benchmark formulation from the literature, which over- (or under-) predicts the densimetric Froude number, possibly due to the presence of effective parameters and their data set ranges. They have also noted that the application of MT and EPR modelling is challenging due to the different ranges of output data used for training and design of sewer networks, even if all the analyzed inputs are available. If few inputs are available, the only approach that should be adopted is EPR modelling.

Matias et al. (2017) fabricated small-scale drops to investigate the effect of their type and height, and the effect of the downstream depth and flow rate on the amount of re-aeration of wastewater and hydrogen sulfide gas release from wastewater. They reported that the use of immersed pre-manhole drops leads to an 82% reduction of the re-aeration coefficient compared to free-fall drops. Also, non-immersed pre-manhole drops show a 54% reduction in the re-aeration coefficient compared to free-fall drops. The results of this research can be used in the design of new sewers and in calculating the amount of dissolved oxygen within sewage networks. One of the effects of creating waterfalls in sewers is the increase in the release of methane and sulfur dioxide in the network, which have negative impacts and should therefore be considered.

Najafzadeh & Bonakdari (2017) used a neuro-fuzzy-based group method of data handling (NF-GMDH) network for predicting velocity limited by depositions within the sewer pipes. They developed the NF-GMDH network using a PSO algorithm for the training stage. Their findings showed that the NF-GMDH-PSO model produced results with relatively lower errors of root mean square error (RMSE) and BIAS than the traditional equations. They tried to predict wastewater speed impacted on by sewer depositions using sediment characteristics and hydraulic properties, but the prediction of sewer flow velocity in other situations is not possible using this model.
(Carrera et al. 2017a, 2017b) assessed hydrogen sulfide gas transfer from the liquid phase to the gas phase by studying the relationship between the mass transfer coefficient, the airflow turbulence and the water flow. The oxygen mass transfer coefficient \( K_{L,O_2} \) was not affected by the airflow velocity in the range of 0.55–2.28 m/s and water flow velocity in the range of 0.55–0.6 m/s. Finally, they predicted the mass transfer coefficient of hydrogen sulfide \( (K_{L,H_2S}) \) in gravity sewer pipes under similar hydraulic conditions using the ratio of \( K_{L,O_2} \) to \( K_{L,H_2S} \). The researchers studied the mass transfer coefficient of hydrogen sulfide under laminar flow conditions and did not make any recommendations for transient and turbulent flow conditions as well as structures such as drops where more gas is released. They also used the oxygen mass transfer coefficient to predict the hydrogen sulfide mass transfer coefficient, which is likely to be challenging.

Carrera et al. (2017a, 2017b) conducted studies to measure changes in the mass transfer coefficient under hydraulic conditions with the aim of calculating the amount of hydrogen sulfide emissions in a gravity sewer. They collected data to develop models of wastewater quality changes using system geometry, and they tested the ratio between the mass transfer coefficients of \( H_2S \) and \( O_2 \) in a complete mixing reactor of 8 l under different conditions. Then, they examined the oxygen transfer coefficient in a 10-m-long gravity pipe in the speed range of 0.027–0.61 m/s and the Reynolds number range of 4.332–46.130. Results of this study showed that in the laboratory-scale reactor, the oxygen mass transfer coefficient \( (K_{L, O_2}) \) depends on Equation (1) to the stirring speed \( N \) (rph):

\[
K_{L,O_2} = 0.016 + 0.025N^{3.85} \tag{1}
\]

where \( K_{L,O_2} \) is the oxygen mass transfer coefficient (m h\(^{-1}\)), and \( N \) is the stirring speed inside the reactor (h\(^{-1}\)).

They also established the relationship between the oxygen mass transfer coefficient \( K_{L,O_2} \) in sewer pipes according to Equation (2) with the flow velocity in the pipe \( U_i \) (m h\(^{-1}\)).

\[
K_{L,O_2} = 0.016 + 1.02 \times 10^{-5}U_i^{3.85} \tag{2}
\]

They reported an exponential increase of the oxygen mass transfer coefficient \( K_{L,O_2} \) with increasing mean flow velocity, which indicates that the increase in dissolved oxygen transfer is proportional to the increase in flow turbulence.

Heydarzadeh et al. (2019) presented a sewage network design model to enhance the organic matter removal efficiency of the sewers. They optimized the objective functions by creating a model for calculating hydraulic and qualitative parameters of the sewers. They found that removal efficiency could be increased up to 30% but sometimes at the price of deteriorating some other network functions. Thus, to perform this optimization, it is important to follow stakeholder indications. All experiments were intended to determine the COD parameters and performed in aerobic and steady-state flow conditions. They performed their study under aerobic conditions, steady-state flows, and only for COD parameter. So, their findings did not account for anaerobic conditions, variable flows and other wastewater quality parameters.

Sotelo et al. (2020) investigated the impact of flow frequency on the degradation of fatty matter by examining the oxygen consumption rate, the fat consumption evolution and the concentration of dissolved oxygen in a small spongy medium, and they concluded that the water flow favours oxygen mass transfer to the spongy medium. They measured the oxygen consumption rate in aerobic conditions for fatty matter degradation and biofilm activity as well as the re-aeration in the absence of flow and for flows in the range of 0.41–1.52 kg O/m\(^3\) d. At alternating flow conditions and high dissolved oxygen concentrations, C16 fatty acid degradation is preferred over C18 fatty acid degradation. Consequently, continuous flow is generally an advantage as it leads to an increase of wastewater treatment even though in real sewers, this condition usually does not occur. In this study, no difference has been determined between the role of biofilms and biological suspended solids on the degradation of fatty matters.

As a conclusion of all the selected studies, most researchers have investigated the effect of one or more parameters on the wastewater within sewers. Because all mentioned studies have been performed on pilot-scale or small sewer networks, the variation of the networks’ dissolved oxygen concentrations under different hydraulic conditions and various wastewater qualities within a large sewer network have not been addressed.

For this research, our aim was to model an individual gravity sewer network with specific physical characteristics such as a constant roughness coefficient and assumed uniformity. The objectives were to (a) allow for model calculation of the dissolved oxygen concentration related to all key factors affecting it and (b) easily determine the values of the dissolved oxygen concentration within a large sewer network.
oxygen concentration at different positions within the wastewater collection network for various different hydraulic conditions.

**Study area**

The performance of the written qualitative model for dissolved oxygen should be examined in a real sewer network with multiple inputs and sewer branches. For this purpose, the western part of Birjand city sewage network with a length of 50 km, including 995 manholes and 995 double-walled polyethylene pipes with diameters of 200, 250, 315, 400 and 500 mm, was selected. The required information was extracted from the geographic information system map of the studied sewage network consisting of shape files of manholes, pipes, water subscribers and urban pathways. The shape files were constructed by taking the surveyed length, slope, depth and diameter of the pipes, their geographical location, their diameter, the depth of the manholes as well as the locations of the water branches in the studied area into consideration. The model input required the calculation of the wastewater quality changes per capita. Quality parameters of the discharged wastewater such as COD and dissolved oxygen as well as hydraulic Gauckler–Manning–Strickler pipe coefficients were also determined.

The studied sewage network is represented in Figure 1. To calibrate and validate the dissolved oxygen model, flow rate and dissolved oxygen in different positions of the sewer network were estimated. For this purpose, a series of corresponding sampling locations were selected. The criteria for selecting them were as follows:

1. There should be sufficient flow in the sewer pipeline, so that it is possible to measure the height of the flow.
2. It should be possible to sample the wastewater to measure dissolved oxygen.
3. The selected locations should be on the main pipes, if possible, so that the changes in the sewage after connecting the side lines as well as structures such as drops can be evaluated.
4. The distribution of locations in the network should be uniform, so that the whole network is covered.

According to the selection criteria of sampling points, eight points named 1C297, 1C449a, 1C676, 1C807, 1B51, 1B128, 1B225 and 1B823 were selected on the main lines according to Figure 1. Sewage sampling was performed to determine dissolved oxygen and flow height in the pipes according to Table 1. The measured values were then compared with the values obtained from the model.

Owing to the need for the same hydraulic conditions in the model and in the studied sewer network, the flow height in one of the main sewer pipes of the whole study area (manhole 1b823 located at the outlet of pipe 1b822t1b823) is measured and the effluent flow in the pipe is determined using slope, pipe diameter and the Gauckler–Manning–Strickler formula. The wastewater volume for each of the 7,362 sewer branches per capita was calculated. The effluent discharge for each manhole, secondary sewer pipe and main sewer pipes were also estimated based on the corresponding density of their occurrences. However, the non-simultaneous production of wastewater by water subscribers as well as the difference in the amount of wastewater they produce leads to variability in the estimated discharge and the measured wastewater amount within pipes.

To determine the COD of wastewater discharged into the network during the measurement period of flow height and dissolved oxygen at the locations specified in Figure 1, samples were taken from four different points of the studied network branches. The average COD for the four points was 676, 646, 643 and 671 mg/l, respectively, and their average of 660 mg/l was entered into the model as the input COD of the wastewater into the network. Although the concentration of dissolved oxygen in the branches varied between 1 and 1.5 mg/l, its average of 1.2 mg/l was used as an input for the model as the dissolved oxygen of wastewater discharged to the network by subscribers.

To calculate the quality changes of wastewater, the percentages of $S_S$, $X_S$ and $X_H$ must be experimentally determined. However, due to the impossibility of performing COD tests during sampling, and according to the literature, $S_S$, $X_S$ and $X_H$ were set at 28, 32 and 5% in that order (Hvitved-Jacobsen et al. 2013). Owing to the calibration of the qualitative parameters of the model, the COD component estimation error was largely minimized.

**MATERIALS AND METHODS**

Determining the parameters of wastewater in the collection network requires the use of two models: a hydraulic model and a model for determining the wastewater quality. In this section, the computational models and their applications are introduced.
Figure 1 | Map of the western part of the Birjand city sewage network and sampling points.
Hydraulic model

To model wastewater qualitative changes, some hydraulic parameters of the network such as the cross-sectional area of the flow, flow velocity, flow rate and wetted perimeter for each pipe are needed. The use of established hydraulic models such as SewerCAD and SWMM was not possible. Within the SewerCAD software, it is not possible to calculate wastewater quality changes. Furthermore, it is not an open-source software, so it cannot be connected to a model that calculates quality changes. The SWMM can estimate the production of pollutant loads associated with storm water runoff. It generates non-point source pollutant loads for waste load allocation. While in sewer networks, researchers are faced with point source pollutants. It is also not easy to change the text of the SWMM program written by different people. Therefore, it makes sense to write a program to simultaneously calculate the hydraulic and quality parameters of the wastewater collection network and verify its performance. Therefore, a program was written in the C++ programming environment to perform the hydraulic calculations on the network. The flow rate of each pipe was determined from the total flow rate of the upstream pipes and the sewage discharged by the surrounding subscribers to the manhole at the beginning of that pipe. The slope and diameter of the pipe were determined using mapping. Assuming a uniform flow, applying the continuity equation and the Gauckler–Manning–Strickler formula, hydraulic parameters of the network are determined by the program.

To verify the validity of the program, the authors simulated a network consisting of four pipes with three different discharges using SewerCAD as well as the written program. A comparison of the results of the two models showed that the difference between the calculated values of the flow height \(Y\) and its velocity \(V\) using the written model and SewerCAD was \(<2\%\). Also, the correlation coefficient \(R^2\) for the flow height \(Y\) and velocity \(V\) was approximately equal to one, which indicated a good correlation between the results of the two programs. It should be noted that if the continuity equation prevails, the validation of the model will have the same results at different conditions. Considering that the model performs hydraulic calculations using the Gauckler–Manning–Strickler formula, as long as the values of the parameters are entered correctly, the result should be sufficiently accurate.

Modelling quality changes

Usually, the quality of domestic wastewater changes significantly, while it flows in the sewage network, mainly because of the reduction of biodegradable materials and the production of biomass (Bjerre et al. 1998; Abdul-Talib et al. 2002; Hvitved-Jacobsen et al. 2013). Changes in wastewater components are complex as they are the result of many simultaneous processes. The microbial processes that govern wastewater quality are actually biochemical changes that take place in the biological environment and are carried out by living organisms. The main factor influencing the progress of biochemical reactions is the chemical conversion energy. Microorganisms use food for growing and retrieve energy from their biomass through oxidation and electron transfer (Hvitved-Jacobsen et al. 2013). The characteristics of the wastewater within sewer systems

Table 1 | Height of flow and dissolved oxygen in the western part of the Birjand sewage network

| Date          | Time  | Manhole | Height of flow (cm) | Dissolved oxygen (mg/l) | Wastewater temperature (°C) |
|--------------|------|---------|--------------------|-------------------------|-----------------------------|
| 12 December 2018 | 10:43 | 1B128   | 11.5               | 2.10                    | 25                          |
|              | 11:14 | 1B225   | 10.8               | 2.15                    | 25                          |
|              | 10:30 | 1B51    | 5.5                | 3.30                    | 25                          |
|              | 12:10 | 1C297   | 5.5                | 3.65                    | 25                          |
|              | 10:20 | 1C449a  | 15.5               | 3.80                    | 25                          |
|              | 10:58 | 1C676   | 15.8               | 1.45                    | 25                          |
|              | 11:50 | 1B823   | 20.8               | –                      | –                           |
|              | –     | 1C807   | –                  | –                      | –                           |
| 26 December 2018 | 10:12 | 1B128   | –                  | –                      | –                           |
|              | 1B225 | 7.0     | –                  | –                      | –                           |
|              | 1B51  | –       | –                  | –                      | –                           |
|              | 1C297 | –       | –                  | –                      | –                           |
| 09:05        | 1C449a| 10.5    | –                  | –                      | –                           |
| 10:00        | 1C676 | 9.0     | –                  | –                      | –                           |
| 10:30        | 1B823 | 17.5    | –                  | –                      | –                           |
| 10:46        | 1C807 | 12.3    | –                  | –                      | –                           |


depend on factors such as the type of sewer, the network retention time, the temperature as well as the presence and type of industrial wastewater discharged into the network (Almeida 1999). There are different phases inside the sewer pipes:

1. A liquid phase includes soluble and suspended substances.
2. The air or gas phase between the liquid phase and the pipes’ inner surface does not exist in pressurized pipes.
3. The biofilm is attached to the pipe’s inner walls and sediment’s surface.
4. The sediments accumulate in the bottom of the pipe.

The presence of microorganisms in different phases of wastewater including the liquid, the sediment and the biofilm phases is the driving force behind the biochemical processes. Heterotrophic biological mass uses organic matter as a carbon source for microbial growth as well as an energy source for processes related to biomass growth (Hvitved-Jacobsen et al. 2013).

Different sewers can be divided into two categories: the gravity sewers and the pressurized ones. In gravity networks, aerobic processes are predominant because of longer retention times and also because of the aeriation of wastewater made possible due to the presence of the gas phase, and the removal efficiency of organic matter is higher than in pressurized sewers. Concerning the pressurized sewers, biofilms and anaerobic processes are more effective in removing organic matter but have lower removal efficiencies than gravity sewers.

Aerobic, anaerobic and anoxic processes take place inside the sewers, depending on the type of wastewater electron acceptor, including oxygen, sulfate and nitrate. Dissolved oxygen is a key factor for aerobic conditions. The main aerobic sewers are very important in situations where the wastewater transition time is long, because the quality of the wastewater changes significantly during transport. Under such conditions, the amount of biodegradable substrate is reduced and biomass is produced. The aerobic conversion of organic matter mainly depends on three factors: dissolved oxygen as the electron acceptor, biodegradable organic substrate as the electron donor and heterotrophic microorganisms (Hvitved-Jacobsen et al. 2013).

The concentration of organic matter in the sewers depends on factors such as the organic load per capita, water consumption rate, collection network runoff and presence of industrial and commercial areas. The typical amount of each COD component within a sewer can be estimated using Equation (3). However, their exact value must be found out by testing (Hvitved-Jacobsen et al. 2013).

\[
\frac{S_F}{3} = \frac{S_f}{3} = \frac{X_{S_f}}{14} + \frac{X_{S_f}}{70} = \frac{X_{HF}}{10}
\]

where \(S_F\) is the readily (fermentable) biodegradable substrate \((\text{gO}_2 \text{m}^{-3})\); \(S_A\) is the volatile acid or fermentation product \((\text{gO}_2 \text{m}^{-3})\); \(X_{S_f}\) is the fast hydrolyzable organic matter \((\text{gCOD m}^{-3})\); \(X_{S_f}\) is the slow hydrolyzable organic matter \((\text{gCOD m}^{-3})\) and \(X_{HF}\) is the heterotrophic biomass concentration in the water phase \((\text{gCOD m}^{-3})\).

The wastewater aerobic/anaerobic transformations in sewers (WATS) model was applied to simulate the aerobic changes of organic matter as well as the re-aeration in the wastewater collection network. WATS is also a suitable conceptual model for estimating wastewater organic matter in both the liquid and biofilm phases under different conditions such as aerobic and anaerobic ones (Hvitved-Jacobsen et al. 2013). The authors used the concept of WATS in their model for calculating wastewater quality parameters.

To use the WATS, the mass equilibrium equations of different components of the organic matter were written and coded in C++. The equations used for modelling aerobic conditions in the WATS model depended on the COD main component variation in the liquid phase over time as well as the evolution of the dissolved oxygen concentration in the sewers (see Equations (4)–(7) and also refer Almeida (1999)).
\[ \frac{dX_s}{dt} = K_{hn} \frac{X_{Sn}}{X_{HW}} \frac{S_O}{K_S + S_O} \left( X_{HW} + eX_{HF} \frac{A}{V} \right)^{\alpha_W (T-20)} \]  

(5)

\[ \frac{dX_{HW}}{dt} = \mu_{H,O} \frac{S_F + S_A}{K_{SW} + (S_F + S_A)} \frac{S_O}{K_O + S_O} X_{HW}^{\alpha_W (T-20)} + K_{18}/5 \frac{Y_{HW}}{1 - Y_{HW}} \frac{S_F + S_A}{K_S + (S_F + S_A)} \frac{A}{V}^{\alpha_W (T-20)} - q_m \frac{S_O}{K_O + S_O} X_{HW}^{\alpha_W (T-20)} \]  

(6)

\[ \frac{dS_O}{dt} = K_{La} (S_{Os} - S_O) - \left( \frac{1 - Y_{HW}}{Y_{HW}} \right) \mu_{H,O} \frac{S_F + S_A}{K_{SW} + (S_F + S_A)} \frac{S_O}{K_O + S_O} X_{HW}^{\alpha_W (T-20)} - K_{SO}^{0.5} \frac{S_F + S_A}{K_{SF} + (S_F + S_A)} \frac{A}{V}^{\alpha_W (T-20)} \]  

(7)

where \( S_s \) is the biodegradable substrate (gO\(_2\) m\(^{-3}\)); \( \alpha_W \) is the temperature coefficient for water-phase processes; \( \mu_{H,O} \) is the maximum specific growth rate (d\(^{-1}\)); \( e \) is the relative efficiency constant for hydrolysis of the biofilm biomass (-); \( K_{hn} \) is the hydrolysis rate constant for fraction \( n \) (d\(^{-1}\)); \( K_O \) is the saturation constant for dissolved oxygen (gO\(_2\) m\(^{-3}\)); \( K_{SW} \) is the saturation constant for readily biodegradable substrate (gCOD m\(^{-3}\)); \( K_{SF} \) is the saturation constant for readily biodegradable substrate within a biofilm (gCOD m\(^{-3}\)); \( K_{sn} \) is the saturation constant for hydrolysis of fraction \( n \); \( q_m \) is the maintenance energy requirement rate constant (d\(^{-1}\)); \( Y_{HW} \) is the liquid-phase yield constant for biomass (gCOD, biomass (gCOD, substrate)\(^{-1}\)); \( Y_{HF} \) is the biofilm yield constant for biomass (gCOD, biomass (gCOD, substrate)\(^{-1}\)); \( Fr \) is the Froude number (-); \( u \) is the velocity of the flow (ms\(^{-1}\)); \( s \) is the slope of the pipe (mm\(^{-1}\)); \( d_m \) is the mean depth of the flow in the pipe (m); \( S_O \) is the dissolved oxygen concentration in bulk water phase (gO\(_2\) m\(^{-3}\)); \( S_{OS} \) is the saturation dissolved oxygen concentration in the bulk water phase (gO\(_2\) m\(^{-3}\)); \( K_{1/2} \) is the 1/2-order rate constant (O\(_2\)\(^{0.5}\) m\(^{-0.5}\)d\(^{-1}\)); \( X_{HW} \) is the heterotrophic biomass concentration in the water phase (gCOD m\(^{-3}\)); \( A \) is the biofilm surface area (m\(^2\)) and \( V \) is the volume of wastewater in the pipe (m\(^3\)).

In Equation (7), \( K_{La} \) is the re-aeration coefficient for wastewater (d\(^{-1}\)). It is calculated using Equation (8) (Huisman et al. 1999).

\[ K_{La} = 0.66(1 + 2Fr^2)(s/u)^{3/8}d_m^{-1} \]  

(8)

Increasing the slope in Equation (7) leads to an increase of velocity, reduction of the flow depth and an increase of the Froude number, thus increasing the amount of wastewater aeration. Also, the amount of wastewater aeration caused by the drops is calculated by the following equation (Hvitved-Jacobsen et al. 2013).

\[ r_O = \frac{S_{OS} - S_{u,O}}{S_{OS} - S_{d,O}} \]  

(9)

where \( S_{u,O} \) is the dissolved oxygen concentration upstream of the drop (g m\(^{-3}\)); \( S_{d,O} \) is the dissolved oxygen concentration downstream of the drop (g m\(^{-3}\)) and \( r_O \) is the ratio of dissolved oxygen deficiency for aeration in the drop, which is determined by Equation (10) as indicated by Matos & Sousa (1991).

\[ r_O = e^{0.45H^0.125H^2} \]  

(10)

where \( H \) is the drop height (m).

Owing to the large number of unknown variables, it is not possible to solve Equations (5)–(8) using mathematical analytical methods and therefore they must be solved using numerical methods. For that purpose, the fourth-order Runge–Kutta Method was performed due to its sufficient accuracy and stability of the results in solving the selected equations.

Equations (4)–(7) are used for complete mixing reactors. If they are applied for a pipe, they cause some instability in the program. To avoid this challenge, the pipe can be divided into smaller sections with \( \Delta X \) length and each section can be considered as a complete mixing reactor (Gujer & Wanner 1989). The mass survival equations for each reactor are formulated and solved according to Equations (11) and (12). The output concentration of each reactor is equal to the input concentration...
of the next one

\[
\text{Input-Output} + \text{Reaction} = \text{Accumulation} \tag{11}
\]

\[
C_{i,j-1} Q_{i,j-1} - C_{i,j} Q_{i,j} + r V_{i,j} = (C_{i,j} V_{i,j} - C_{i,j-1} V_{i,j-1}) / \Delta t \tag{12}
\]

where \( C_{i,j} \) is the concentration in cell \( i \) at time \( j \); \( Q_{i,j} \) is the flow output from cell \( i \) at time \( j \); \( V_{i,j} \) is the volume of cell \( i \) at time \( j \); \( r \) is the reaction rate and \( \Delta t \) is the time step.

Owing to the rather long retention times in each reactor and the variability of wastewater, the time step in solving Equations (4)–(12) was defined, so that in each time step the equations are solved and new concentrations are calculated. This leads to more accuracy and stability of the data. Based on the proposal by Hvitved-Jacobsen et al. (2013), a longitudinal step of 20 m and a time step of 10 s were considered. Finally, after solving Equations (4)–(12) in time and longitudinal steps, the changes in chemical oxygen concentration in each pipe are obtained from the sum of its changes in the assumed reactors and are noted according to Equations (13) and (14) according to Almeida (1999).

\[
\text{COD}_{\text{removal pipe}} = \sum \Delta S_i + \sum \Delta X_s - \sum \Delta X_{HW} \tag{13}
\]

\[
\text{COD}_{\text{out pipe}} = \text{COD}_{\text{in pipe}} - \text{COD}_{\text{removal pipe}} \tag{14}
\]

The concentration of dissolved oxygen at the outlet of each pipe is calculated based on the following equation:

\[
\text{DO}_{\text{out pipe}} = \text{DO}_{\text{in pipe}} + \sum \Delta S_0 \tag{15}
\]

where \( \text{DO}_{\text{in pipe}} \) is the concentration of dissolved oxygen at the inlet of the pipe and \( \text{DO}_{\text{out pipe}} \) is the concentration of dissolved oxygen at the outlet of the pipe.

In our program, Equations (4)–(7) are solved by the fourth-order Runge–Kutta Method, and the COD concentration within the pipes is based on Equations (13) and (14), while dissolved oxygen is determined according to Equation (15). Additionally, the qualitative changes of the sewage at the intersections of the pipes are calculated using the weighted average, which is based on the volumetric flow rate.

The model is partly based on Equations (4)–(6) used to calculate wastewater work changes. The condition of the wastewater entering the network is known and fixed. The initial conditions for the first set of pipes are the same conditions as for sewage entering the network. In the next step, the conditions of the effluent from each pipe are the initial conditions of the next pipe.

**RESULTS AND DISCUSSION**

**Model calibration**

The model must be hydraulically and qualitatively calibrated to ensure its proper operation. In the hydraulic calibration, the sole variable parameter is the Gauckler–Manning–Strickler coefficient \( n \). Chow (1959) reported \( n \) values for sewage networks ranging from 0.012 to 0.016. Additionally, in 2007, the American Concrete Pipe Manufacturers Association proposed \( n \) values ranging from 0.009 to 0.015 for high-density polyethylene (HDPE) sewer pipes. Devkota et al. (2012) obtained a value of \( n \) of 0.011 for sewage polyethylene pipes with a diameter of 300 mm and a filling ratio of 45%.

The physical parameters such as diameter, length and slope of the studied network pipes were specified, but the precise values of \( n \) were not available for them. So, the only option available was to change and calibrate the model hydraulically with different values of \( n \). As the pipe material was double-walled corrugated polyethylene and the pipe age range was between 10 and 15 years, the same coefficient was assumed for the whole network. According to the values proposed in the literature, the hydraulic calibration of the network was performed assuming the values of 0.011, 0.013, 0.014 and 0.015 for \( n \). For the hydraulic and qualitative calibration of the model, dissolved oxygen and the flow height in one of the main wastewater collection pipes (manhole 1B823 located at the outlet of pipe 1B822(1B823) and in the other six points of the network including the manholes 1B128, 1B225, 1B51, 1C297, 1C449a and 1C676 (Figure 1) were measured. 1C807 was removed from the sampling list due to its proximity to the confluence of two sewage lines and the absence of streams at that point, which led to an error in measuring dissolved oxygen. Using the slope and pipe diameter, and assuming a value of \( n \) for the whole network and applying the Gauckler–
Manning–Strickler formula, the effluent flow in the pipe 1B822t1B823 with manhole 1B823 was calculated. Then, the per capita wastewater production for each water branch was estimated using the flow rate calculated for the pipe 1B822t1B823.

Finally, the wastewater flow rate and height in the main and sub-pipes including six sampling points were calculated by the model. As wastewater re-aeration in the pipe is affected by the Froude number of the flow, which is also dependent on the pipe coefficient \( n \), in addition to the flow height measured in the pipe, the concentration of dissolved oxygen within the wastewater was measured at specific locations to calibrate the value of the network coefficient \( n \). Therefore, the sum of squares of errors (SSE), \( R^2 \), the values obtained from the model for the flow height in the pipe and the concentration of dissolved oxygen were calculated for different values of \( n \). The value of \( n \), for which the minimum values of SSE and the maximum value of \( R^2 \) (relatively closer to one) were obtained, was determined for the network.

Based on the sensitivity analysis performed by Almeida (1999), it was found that the parameters \( Y_H, \mu_H, K_{OG} \) and \( K_{SF} \) have the highest sensitivity in the equations linked to wastewater quality change. Based on the previous studies, \( \mu_H \) between 3 and 8.5 \( \text{d}^{-1} \), \( Y_H \) between 0.3 and 0.8 \( \text{gCOD}/\text{gCOD} \), \( K_{OG} \) from 0.001 to 1.5 \( \text{gO}_2/\text{m}^3 \) and \( K_{SF} \) in the range of 0.5–1.5 \( \text{gCOD}/\text{m}^3 \) were selected for model calibration purposes by keeping three parameters constant and changing the fourth parameter for the calibration. Concerning the amount of dissolved oxygen measured and obtained from the model, the value that minimizes the sum of square errors and maximizes \( R^2 \) was selected as the optimal value.

The concentration of dissolved oxygen and the height of the flow in the manholes 1B128, 1B225, 1B51, 1B823, 1C297, 1C449a, 1C676 and 1C807 were measured at two different days and the findings are summarized in Table 1. Using the results of the measured height of flow and dissolved oxygen on 12 December 2018 and 26 December 2018, the model was calibrated in terms of hydraulics and water quality. The calibration results showed that with an increasing coefficient \( n \), the values of SSE and \( R^2 \) for the flow height do remain constant because of the small difference in the estimated flow rates for different coefficients of \( n \).

The amount of dissolved oxygen calculated by the model was, however, close to the measured value. The reason for this is the lack of synchronous production of wastewater by water subscribers, which leads to the difference between the estimated flow rate for pipes in the model compared to the measured values. Increasing the coefficient \( n \) by affecting the Froude number reduces the re-aeration of wastewater within pipes and leads to a reduction in the distance between the amount of dissolved oxygen calculated by the model and the measured amount.

Also, considering the 10–15 years of sewer pipe lifespan and the possibility of having sediments within the pipes, the coefficient \( n \) within the network is expected to increase. A value of 0.015 was selected for \( n \), which is in the range recommended in the literature for polyethylene sewer pipes, while the values for SSE and \( R^2 \) calculated for dissolved oxygen are the minimum and maximum values, respectively. In the next step, the calibration of \( Y_H, \mu_H, K_{OG} \) and \( K_{SF} \) was performed based on the values extracted from the previous studies and assuming \( n \) to be constant at 0.015.

Owing to the very small difference in the value of \( R^2 \), the minimum value of the SSE was selected for calibration. Finally, \( Y_H, \mu_H, K_{OG} \) and \( K_{SF} \) were set at 0.33, 8.4, 0.6 and 0.75, respectively. The SSE values for the flow height and the dissolved oxygen were 15.5901 and 1.6872, respectively. Also, the values of \( R^2 \) for dissolved oxygen and the flow height were 0.7706 and 0.8214 in this order. According to the SSE values, the RMSE of the flow height and the dissolved oxygen were 1.7658 and 0.7545, respectively.

**Changes in dissolved oxygen in the sewer network during the day**

The volume and quality of wastewater changes during the day and night due to diets and habits of different people worldwide (Almeida 1999). To assess the changes in the concentration of dissolved oxygen within the sewer network, the authors varied the flow rate during the day. Dissolved oxygen concentrations within the network at minimum and maximum flow conditions were simulated by the model. For this purpose, the statistics of inlet flow to the Birjand wastewater treatment plant on 26 December 2018, which was measured every 10 min by the inlet flow measurement unit of the treatment plant, were used and the change curve of the treatment plant inlet flow at different hours of the day was plotted according to Figure 2.

According to Figure 2, the maximum inlet flow of the Birjand wastewater treatment plant occurred at 15:10 and was equal to 330.83 l/s, while the minimum flow was recorded at 07:10 and was of 46.84 l/s. According to Table 1, flow height and dissolved oxygen for manhole 1B823 were recorded at 10:50 on the same day. Owing to the constant flow within different parts of the transmission line and the challenges of measuring flow, the floating body method was used to measure the average speed of sewage and its movement in time. At around 10:50, a plastic ball was thrown into the manhole 1B823 and the time it took to arrive at the treatment plant was recorded at 114 min (12:24). At this time, the inlet flow of the treatment plant
was 286.68 l/s. In this way, the maximum and minimum flow coefficients for the western part of Birjand city sewage network on 26 December 2018 were obtained by dividing the maximum and minimum inlet flows on that day, which equates to 1.15 and 0.14, respectively. By knowing the maximum and minimum flow coefficients, the maximum and minimum discharge conditions were modelled for the western part of the Birjand wastewater collection network and the dissolved oxygen changes were investigated.

Owing to the variability of COD during the day, it is better to measure its average concentration entering the network at different hours, but due to the challenges of measuring COD at different hours, a sensitivity analysis for the changes in the dissolved oxygen model was performed by changing the COD concentration within the network. The model was run with COD values of 500, 550, 600, 660, 700, 750, 800, and 850 mg/l, and the following corresponding average concentrations of dissolved oxygen in the network were obtained from the model: 2.99, 2.89, 2.80, 2.70, 2.64, 2.56, 2.48, and 2.16 mg/l, respectively. If the average value of COD entering the network decreases by 160 mg/l compared to the measurement conditions and reaches 500 mg/l, the average dissolved oxygen of the network increases by 10% and reaches 0.286 mg/l. If the average value of COD input to the network increases by 190 mg/l compared to the measurement conditions and reaches 850 mg/l, the average dissolved oxygen concentration of the network decreases by 0.55 mg/l, which equates to 20%. Therefore, the sensitivity value of the dissolved oxygen change function due to changes in the COD value was calculated based on the Morris method according to the following equation (Morris 1991):

$$S_i = \sum_{j=1}^{r} EE^j = \frac{1}{r} \sum_{j=1}^{r} \frac{g(X_1^j, \ldots, X_i^j + \Delta_i^j, \ldots, X_M^j) - g(X_1^j, \ldots, X_i^j, \ldots, X_M^j)}{\Delta_i^j} C_i$$

where $X_i^j$ is the amount of input to the model; $g(X_1^j)$ is the relationship between the inputs and outputs of the model on which the sensitivity analysis is based; $\Delta_i^j$ is the model input change; $r$ is the sample number and $C_i$ is the scalability factor.

Based on Equation (16), the sensitivity value of the dissolved oxygen function related to COD was 0.00116. For every 1 mg/l change in COD, the dissolved oxygen changes by 0.00116 mg/l, which is a relatively small amount. Therefore, due to the relatively low sensitivity of the average dissolved oxygen concentration in the network, the corresponding average COD of the wastewater entering the network during the day can be assumed to be constant. For the performed measurements, the average dissolved oxygen concentration within the wastewater entering the network in the sub-branches, which always have a very slow flow and constant hydraulic conditions during day and night, was 1.2 mg/l. Therefore, the average dissolved oxygen concentration within the wastewater entering the network can be assumed to be constant.

Results of the dissolved oxygen change model

According to the above sections, the dissolved oxygen concentration within different parts of the wastewater collection network for the three modes of sampling time, maximum flow and minimum flow conditions was calculated using the written quality change calculation program. Given that the wastewater quality change model reports the dissolved oxygen concentration in the manholes on both sides of each pipe, the average of these two numbers was considered as the concentration of dissolved oxygen in that pipe. The dissolved oxygen mean within the sewage collection network of the western part of
Birjand city was also calculated by the weighted average method based on the volume of sewage within the pipes in the three modes of measured, maximum and minimum wastewater production, which were 2.28, 2.19 and 3.42 mg/l, respectively.

In Figure 3, the contour curves of dissolved oxygen in the western part of the Birjand sewerage network in three modes of measurement, maximum and minimum wastewater production are shown using the graphic capability of the SewerGEMS software. As can be seen in Figure 3, the contour curves for dissolved oxygen are very compact due to its small differences at close points, the large number of measurement points and the use of interpolation and extrapolation methods used to draw them. The measured dissolved oxygen concentrations are not much different from those linked to maximum wastewater production. This was due to the proximity of the measurement time to the time of maximum wastewater production.

The lowest average dissolved oxygen concentration within the network for the maximum wastewater volume was equal to 2.19 mg/l, which indicates that the network was not anaerobic. It can be explained by sufficiently steep pipe slopes, high-flow speeds and drops within the network. The dissolved oxygen concentration at minimum wastewater production is approximately 1.5 times that of the maximum wastewater production. This is due to the lower amount of wastewater in the network and consequently less consumption of dissolved oxygen resulting from re-aeration of wastewater. In some parts of the network branches, the dissolved oxygen concentration was about 1 mg/l, which is due to the low speed of wastewater and consequently poor re-aeration capacity within those parts of the pipe network. At the end of the network, the dissolved oxygen concentration decreased to about 1 mg/l. This is due to the relatively high flow of the sewage and the larger filling ratio of the pipe compared to the upper parts of the network, which leads to higher consumption of dissolved oxygen compared to its replacement by the re-aeration process.

![Figure 3](http://iwaponline.com/jh/article-pdf/doi/10.2166/hydro.2021.051/964251/jh2021051.pdf)

**Figure 3** | The contour curve of dissolved oxygen in the western part of the Birjand sewerage network at three modes of (a) measured, (b) maximum wastewater production and (c) minimum wastewater production on 26 December 2018.
CONCLUSIONS

To estimate the dissolved oxygen concentration in different parts of a wastewater network, both a hydraulic model and a model for estimating the wastewater quality parameters should be used. To verify the performance of the hydraulic model, the value of the Gauckler–Manning–Strickler coefficient (n) linked to the network must be specified. For this purpose, the network can be calibrated by measuring the flow rate at different points of the network and its output as well as modelling the flow rate at the same points with different values of the Gauckler–Manning–Strickler coefficient.

The wastewater capita must be calculated using the total measured outflow of the network divided by the number of subscribers. The inflow of sewage into the network can be calculated using the physical position of the subscribers and assumed that their sewage is discharged to the nearest manhole based on capita. To calculate the dissolved oxygen concentration within the network, the quality characteristics (COD and dissolved oxygen) of the wastewater entering the network at different points should be specified. The dissolved oxygen value of the incoming sewage to the sub-branches can almost be constant due to the very low flow rate of these pipes. The COD of the incoming sewage to the network varies during day and night. But it can be assumed to be constant, because the results of a sensitivity analysis that showed changes in the concentration of COD within the sewer network had little impact on the dissolved oxygen concentration within different parts of the network.

To allow for proper performance of the model in terms of calculating wastewater quality change, the COD components including $S_{0}$, $X_{S}$ and $X_{HW}$ should be determined by testing and entered into the sub-model. However, if they are not measured and if the $Y_{H}$, $\mu_{H}$ and $K_{D}$ coefficients are calibrated based on the measured COD values, acceptable results can be obtained from the simulation of wastewater quality changes.

After hydraulic and qualitative calibration of the model, the concentration of dissolved oxygen in different parts of the sewer network can be determined. The main factors affecting dissolved oxygen within the sewage network are wastewater flow rate and the physical structure of the network including slope, diameter, Gauckler–Manning–Strickler coefficient and drop height. Having the exact coordinates of subscriber locations and using this model, the amount of dissolved oxygen within different parts of the network can be calculated with acceptable accuracy by measuring the sewage discharge at the network outlet. Also, by determining the coefficients of flow change at different hours of the day and applying them to the wastewater capita, the dissolved oxygen changes within the network during the day can be calculated.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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