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Spatial impact of organic matters from point sources on stream water quality

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Abstract: To get a better understanding of the spatial impact of organic discharges on stream water quality, a "scenario-testing" approach was proposed to examine how a change in plant treatment efficiency would affect dissolved oxygen (DO) concentration along a stream. An "upper-lower" boundary technique was applied to conduct sensitivity analysis to observe the responses of DO concentration to different DO-related parameters. The results show a non-linear discrepancy of biochemical oxygen demand (BOD) and DO concentration among different treatment scenarios, which indicates the higher the efficiency of the plant treatment, the shorter the time the stream needs to recover. The sensitivity analysis reveals that the larger the amount of the waste discharge, the more sensitive the BOD to the biological removal parameter. In addition, the DO is more sensitive to the biological removal parameter than to the reaeration parameter.
Keywords: waste treatment; dissolved oxygen; biological oxygen demand; scenario testing; sensitivity analysis

1 Introduction

Over the past few decades, surface water, including streams, rivers, lakes and ponds, has experienced a significant decline of water quality due to the discharge of organic waste into water bodies from industrialization, urban expansion, intensive agriculture practices and other anthropogenic activities. For example, coal mine discharges contaminate water and degrade both surface and groundwater quality[1–2]. This degradation is largely caused by the alternation of dissolved oxygen (DO) levels in a water system. A certain amount of DO in water is required to sustain an aquatic ecosystem because most living aquatic species use oxygen in respiration[3]. For example, Trout need DO levels in excess of 8 mg/L, striped bass prefer DO levels above 5 mg/L and most warm-water fish need DO in excess of 2 mg/L[4]. Accordingly, a decrease in DO in surface water can cause fish death and damage of aquatic ecosystem[5].

Organic matters may alter freshwater DO levels by accelerating the biochemical processes of water-borne bacteria. These bacteria use organic matters from wastewater treatment plants and agricultural or urban runoffs as food sources. Bacteria decompose the organic materials while consuming DO. As the amount of bacteria and their food sources increases, so does the amount of their DO consumption, thus reducing DO for fish and other aquatic life. The amount of oxygen that bacteria consume while decomposing organic materials under aerobic conditions is defined as biochemical oxygen demand (BOD)[6]. Apparently, a low BOD indicates water with good quality, whereas a high BOD indicates water with contaminants.

Given the instrumental importance of DO and BOD for surface water quality, it is not surprising that the spatiotemporal dynamics of DO and BOD levels in waterways have become a standard component in most popular in-stream water quality models[7]. For example, in the SIMCAT (simulation catchments), a model developed by UK Environment Agency, one of the key modelling components is to simulate the dynamic relationships between DO and BOD decay, temperature and atmospheric reaeration. In the MIKE-11, a model developed by the Danish Hydraulic Institute and widely used in Europe, the DO sub-model accounts for photosynthetic production, respiration, reaeration, BOD decay and nitrification. The QUAL2E, a popular water quality model developed by the US Environment Protection Agency in collaboration with universities, used a comprehensive sub-model for DO component to incorporate the effects of the algal, nitrogen, phosphorous, atmospheric

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reaeration, sediment oxygen demand and BOD process[8]. Nevertheless, in practice, what factors should be included in the DO sub-model depends on the in-situ pollution conditions[9].

This paper aims to get a better understanding of the impact of changes in wastewater treatment efficiency and modelling parameters on BOD and DO dynamics in the stream waterway. The specific objectives are: 1) to examine the spatial responses of BOD and DO in the stream system to organic waste discharges; and 2) to explore the sensitivity of BOD and DO to the reaeration coefficient and the bacterial removal rate of BOD.

2 Materials and methods

2.1 Study site

A rural stream in a sub-basin with dominant urban-rural land use in the Grand River watershed of Ontario province, Canada, was considered in this study. As a point source, an urban waste treatment plant was located in a stream reach within the watershed. The plant discharged organic waste, which was considered as BOD input to the stream, but with different concentration rates that depended on the designed treatment efficiency.

Before treatment, the raw waste coming from a chemical factory to the treatment plant had an ultimate BOD concentration of 250 mg/L. After the treatment, the concentration BOD_d discharged from the plant depended on the designed treatment efficiency E:

$$BOD_d = 250\times(1-E) \qquad (1)$$

where E is the efficiency in percentage with a value between 0 and 1 (or between 0% and 100%). Once the treated wastewater discharges to the stream, it will mix with the stream water. As a result, the total river flow downstream from the plant will be the sum of the basic stream flow and the effluent from the plant.

Given the pollution concerns in the watershed, wastewater engineers were interested in knowing the BOD and DO levels in stream water at a certain downstream position to aid the decision-making on the design of the treatment efficiency. Such knowledge might be derived from simulating the spatial dynamics of BOD and DO in the stream.

2.2 BOD and DO models

Streeter and Phelps (1925) established the fundamental linkage of BOD and DO profiles to organic waster discharge to streams[10]. Since then, their work had been extended to incorporate more sources and sinks of DO and BOD[11]. The dominant processes that controlled the DO balance in the river were: 1) oxygen demand by carbonaceous and nitrogenous wastes and by bottom deposits; 2) chemical oxygen demand (COD); 3) plant respiration; 4) plant photosynthesis; and 5) the oxygen gained from atmospheric reaeration[8].

We used the Streeter and Phelps' (1925) BOD equation, but extended it to include BOD loss by sedimentation. The BOD concentration downstream from the treatment plant was simulated by the following equation:

$$\mathrm{BOD}\,(t) = \mathrm{BOD}\,(0)\,\mathrm{e}^{-k\,t} \qquad (2)$$

where BOD(t) (mg/L) is the BOD concentration at time t in the water, BOD(0) (mg/L) the initial BOD in the stream (e.g. at the plant after mixing waste discharges with streamflow), t (day) the time for the waste travelling from the plant to downstream point of interest, and k (day^{-1}) the rate constant for BOD degradation.

Furthermore, t could be linked to stream distance through the expression, $t = x/v$, where x was the distance downstream from the plant, and v was the streamflow velocity. The constant k was calculated as the sum of different types of BOD loss, such as bacterial consumption k_1 and loss by sedimentation k_3.

The DO balance equation was represented as the rate of DO change over time[12–13]:

$$\frac{\mathrm{dDO}\,(t)}{\mathrm{d}t} = k_2[C_S - \mathrm{DO}\,(t)] - k_1\mathrm{BOD}\,(t) \qquad (3)$$

where DO(t) is the instantaneous concentration of dissolved oxygen, k_2 the reaeration coefficient, and C_S the BOD saturation concentration at the given water temperature.

2.3 Model parameters and assumptions

Model parameters used in this study were determined based on the principles or numbers suggested in the appendixes of Tchobanoglous and Schroeder (1985) or other literature[3,12,14–15]. The velocity of the stream above and below the plant was assumed to be constant at 5 km/day. The average streamflow was assumed to be 5×10^6 L/day with an average temperature of 10 °C. Also, the plant constantly discharged 5×10^5 L/day of effluent. The BOD decay rate was assumed as k_1=0.23/day at 20 °C for biological removal rate and k_3=0.04/day for loss by sedimentation. For the reaeration coefficient, k_2=0.28/day at 20 °C. However, both k_1 and k_2 were temperature-dependent parameters. They were estimated with the following empirical formula, respectively[14].

$$k_1(t) = 0.23\times1.047[T(t)-20] \qquad (4)$$

$$k_2(t) = 0.28\times1.025[T(t)-20] \qquad (5)$$

where $T(t)$ was the water temperature at time t.

2.4 Simulation scenarios

Model runs were performed with three scenarios through the treatment efficiency (E) at 25%, 50% or 75% to examine the spatial responses of BOD and

DO levels in the stream to the amount of organic waste discharges. Simulations of BOD and DO in the stream were carried out continuously with the time increment (DT) of 0.2 day for a year. A summary of the data related to the model runs was listed in Table 1.

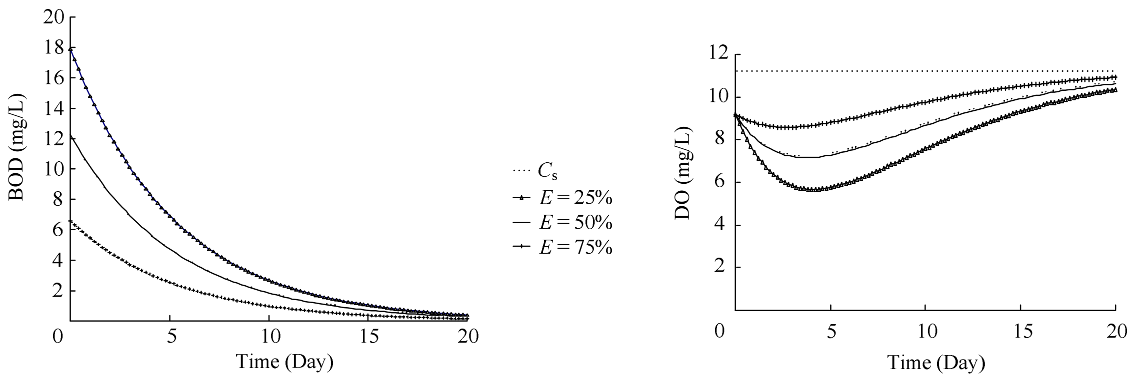
Table 1 Initial data used for scenario testing

E (%)	Plant(1)		Stream(2)		Stream mixture(3)	
	$BOD_p(0)$ (mg/L)	$DO_p(0)$ (mg/L)	$BOD_s(0)$ (mg/L)	$DO_s(0)$ (mg/L)	BOD(0) (mg/L)	DO(0) (mg/L)
25	187.5	1.0	1.0	10.0	17.90	9.18
50	125.0	1.0	1.0	10.0	12.27	9.18
75	62.5	1.0	1.0	10.0	6.59	9.18

Note: (1) $BOD_p(0)$ was the BOD concentration of the effluent discharged from the plant to the stream; $DO_p(0)$ was the DO concentration of the effluent. (2) $BOD_s(0)$ was the BOD concentration of the stream before the wastewater input; $DO_s(0)$ was the DO concentration of the stream before mixing with the wastewater. (3) BOD(0) was the initial BOD concentration of the stream after the wastewater input; DO(0) was the DO concentration of the stream after mixing with the wastewater. $BOD(0) = [Q_p \times BOD_p(0)+Q_s \times BOD_s(0)]/[Q_p + Q_s]$ and $DO(0) = [Q_p \times DO_p(0)+Q_s \times DO_s(0)]/[Q_p+Q_s]$[8]. Here, Q_p was discharges of the effluent from the plant; Q_s was streamflow before mixed with the effluent. The saturation DO at the average temperature of 10 °C was 11.28 mg/L. In addition, at the average temperature of 10 °C, k_1 = 0.15/day and k_2 = 0.22/day.

3 Results

A simulation period with 20 days from the model runs is selected for comparison and demonstration purpose. The dynamics of BOD and DO versus time are shown in Fig. 1.



(a) BOD (b) DO

Fig. 1 Stream BOD and DO dynamics under different waste discharge scenarios from the plant (C_s: saturation DO at the average temperature of 10 °C)

As shown in Fig. 1a, the general declining trends of BOD over time for the three treatment scenarios are similar one with the other. However, it takes different times for the stream to recover the BOD concentration to below 5 mg/L, with approximately 1.5 days for the treatment efficiency at 75%, 5 days at 50% and 7 days at 25%. These times are in corresponding to approximately 8, 25 and 35 km downstream from the plant, respectively. Apparently, the higher the efficiency of the plant treatment, the shorter the time the stream needs to recover. In other words, the higher the efficiency of the plant treatment, the shorter the length of time the stream experienced pollution.

Likewise, the general downward and upward patterns of DO over time for the three treatment scenarios are similar (Fig. 1b). However, the minimum of DO occurs at different times for the three scenarios, at approximately 2.5 days for the efficiency of 75%, 3.5 days for 50% and 4 days for 25%. Furthermore, it takes different times for the stream to recover the DO levels to its original level of 10 mg/L (before waste discharge), with about 11.5 days for the efficiency of 75%, 15.5 days for 50% and 18 days for 25%. These times correspond to approximately 60 km, 80 km and 90 km along the stream from the plant, respectively. Evidently, the higher the efficiency of the plant treatment, the shorter the time the stream needs to recover.

4 Sensitivity analysis

Sensitivity analysis in a modelling context is a technique for detecting the quantitative response of one model output to a small change in a particular parameter[16]. It is often used to explore the role that one single parameter plays in leading to a certain modelling result. For this reason, sensitivity analysis generally yields a sensitive rank of parameters involved in modeling processes. This rank highlights the need for caution in data collection and parameter estimation toward the most sensitive parameter.

For modelling BOD and DO dynamics in the stream, the biological removal parameter (k_1) and the surface reaeration rate (k_2) are recognized as key parameters that influences modelling results dramatically[8]. These parameters are associated with many factors such as temperature, pressure, river depth and streamflow. They can be estimated through theoretical, laboratory or field-empirical relationships developed over decades. For example, more than forty methods have been proposed for estimating k_2[8]. Standard tables with parameter values are also available for reference[17]. However, it is somewhat arbitrary for an individual to pick up a value from a recommended parameter range.

Sensitivity analysis for the parameters can be conducted by first perturbing their values in a range with physical meanings and then examining the modeling response. Different perturbing methods may be used in the sensitivity analysis; however, an upper- and lower-boundary approach with intervals of [0.10, 0.20] for k_1 and [0.17, 0.27] for k_2 is applied for demonstration purposes in this study. Fig. 2 shows the sensitivity of BOD to the perturbation on the biological removal parameter (k_1).

As shown in Fig. 2, within each plant efficiency scenario (E=25%, 50%, or 75%), perturbing biological removal parameter does not alter the declining trend of BOD over time. However, it alters the time the stream needs to recover, with the upper-boundary shortening the time and the lower-boundary increasing the time. Table 2 presents a quantitative comparison for the time alteration in terms of stream recovery to BOD concentration below 5 mg/L. Across the three scenarios, the alteration of time (or distance downstream from the plant) decreases when the plant treatment efficiency is increased (Table 2). This finding implies that the larger the amount of the waste discharge, the more sensitive the BOD to the biological removal parameter, highlighting the need for caution in parameter determination when the discharge is larger.

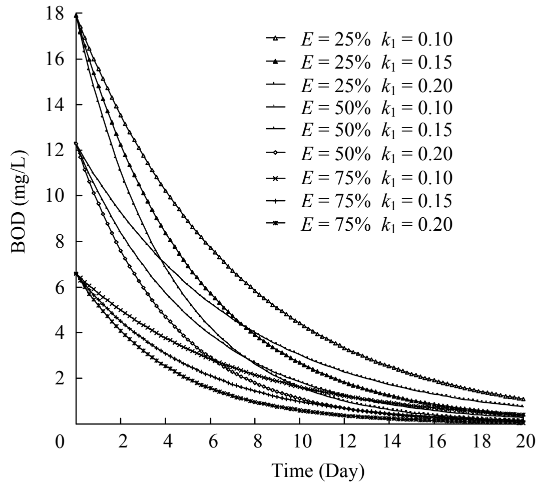


Fig. 2 BOD sensitivity to biological removal parameter (k_1=0.15/day) based on boundaries of [0.10, 0.20] (mg/day)

Table 2 Comparison of stream recovery time to BOD concentration below 5 mg/L

k_1 (mg/L)	E = 25%		E = 50%		E = 75%	
	Time (Day)	Distance (km)	Time (Day)	Distance (km)	Time (Day)	Distance (km)
0.10	9.0	45	6.5	32	2.0	10
0.15	7.0	35	5.0	25	1.5	8
0.20	5.5	25	4.0	20	1.0	5
Δ	3.5	20	2.5	12	1.0	5

Note: Δ-the difference of the time or distance between lower- and upperboundaries.

Table 3 lists the minimum values with their corresponding times. Furthermore, increasing waste discharge would increase the sensitivity of the minimum DO to the biological removal parameters (Table 3). Fig. 3 shows the sensitivity of DO to the perturbation on the biological removal parameter (k_1). Within each plant efficiency scenario (E=25%, 50%, or 75%), perturbing the biological removal parameter neither alters the downward and upward pattern of DO over time nor changes the time when the minimum of DO occurs. However, it alters the minimum value of DO, in that increasing the parameter value would decrease the minimum value (Fig. 3).

Table 4 lists the minimum values with their corresponding times. Furthermore, increasing waste discharge would increase the sensitivity of the minimum DO to the reaeration parameter (Table 4). Fig. 4 shows the sensitivity of DO to the perturbation on the reaeration parameter (k_2). Within each scenario (E= 25%, 50%, or 75%), perturbing the reaeration parameter does not alter the downward and upward pattern of DO over time but slightly changes the time when the minimum of DO occurs. It also alters the minimum value of DO, in that increasing the parameter value would increase the minimum value (Fig. 3).

A comparison between Tables 3–4 and Figs. 3–4 indicates that DO is more sensitive to the biological removal parameter than to the reaeration parameter.

Table 3 Comparison of minimum DO and their occurring times under k_1 perturbation

k_1 (mg/L)	E = 25%		E = 50%		E = 75%	
	Time (Day)	Min DO (mg/L)	Time (Day)	Min DO (mg/L)	Time (Day)	Min DO (mg/L)
0.10	4	6.90	3.5	8.01	2.5	9.01
0.15	4	5.68	3.5	7.16	2.5	8.59
0.20	4	4.69	3.5	6.47	2.5	8.21
Δ	0	2.20	0	1.54	0	0.80

Table 4 Comparison of minimum DO and their occurring times under k_2 perturbation

k_2 (mg/L)	E = 25%		E = 50%		E = 75%	
	Time (Day)	Min DO (mg/L)	Time (Day)	Min DO (mg/L)	Time (Day)	Min DO (mg/L)
0.17	5	4.92	4.5	6.62	3.5	8.28
0.22	4	5.68	4	7.16	3	8.59
0.27	3.5	6.25	3	7.57	2	8.81
Δ	1.5	1.33	1.5	0.95	1.5	0.53

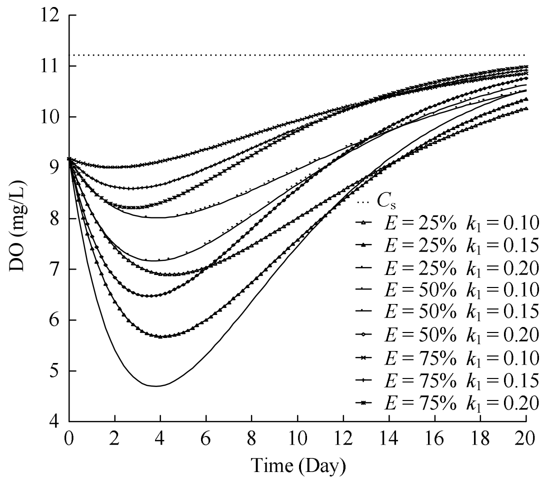


Fig. 3 DO sensitivity to the biological removal parameter (k_1=0.15/day) based on boundaries of [0.10, 0.20] (mg/day)

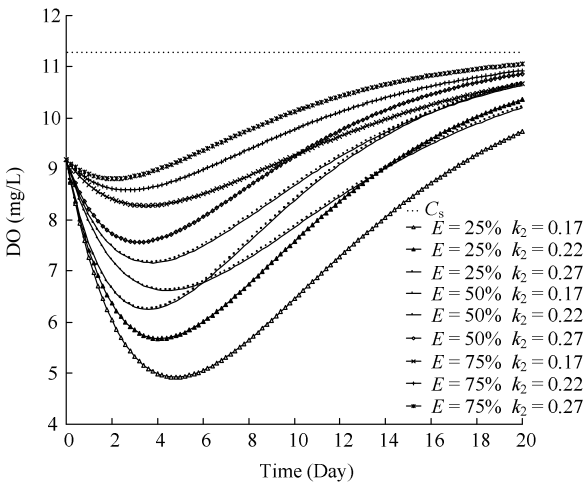


Fig. 4 DO sensitivity to the reaeration parameter (k_2=0.22/day) based on boundaries of [0.17, 0.27] (mg/day)

5 Conclusions

A non-linear discrepancy of biochemical oxygen demand (BOD) and DO concentration exhibits in the stream among different treatment scenarios. The higher the efficiency of the plant treatment, the shorter the time the stream needs to recover. Moreover, the larger the amount of the waste discharge, the more sensitive the BOD to the biological removal parameter. The minimum value of DO is also sensitive to the biological removal parameter. Increasing waste discharge would increase the sensitivity of the minimum DO to the biological removal parameter. Similarly, increasing waste discharge would increase the sensitivity of the minimum DO to the reaeration parameter. However, DO is more sensitive to the biological removal parameter than to the reaeration parameter.

The baseline for further research has been established in this study. First, constants such as streamflow and velocity in the model used here can be adjusted to account for weather and seasonal changes. Second, more physical processes such as photosynthesis, nitrogen cycle and phosphorous cycle can be incorporated into the model to simulate in-stream water quality in a comprehensive manner. Third, the model may be linked to a hydrological model for other watershed studies.

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