Simulating transformation of nitrogen components in sewer system when oxygen and flow velocity changed

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Abstract: In this study, a mathematical model based on the kinetic of Activated Sludge Model was established to describe the transformation of nitrogen compounds including nitrate and nitrite nitrogen, ammonia and ammonium nitrogen, soluble biodegradable organic nitrogen, and particulate biodegradable organic nitrogen in sewer. Then the effects of varied flow conditions and different initial dissolved oxygen (DO) concentrations on nitrification and denitrification were explored. The results showed that these four compounds were transformed during the transportation process. According to simulation, when DO varied and flow velocities were fixed, all nitrogen compounds varied slightly excepting nitrite and nitrate nitrogen. When initial DO was fixed, the different reaction time due to different flow velocity affected the nitrogen transformation significantly. Removal of nitrogen compounds was better when reaction time was greater.

Keywords: Sewer system; mathematical model; nitrogen; aerobic; anaerobic.

Introduction

Collection and treatment of municipal wastewater become more important in Taiwan because the amount of wastewater generated from residential and business sectors is increasing year by year with the expansion of population. But it encounters a challenge because of the low coverage of sewer system in Taiwan, only 15.58 % by 2006. The Executive Yuan announced that the coverage will be promoted to 20% by 2008.

Because the sewer systems receive large amounts of wastewater from either industrial effluents or households with high concentrations and various compounds including many nitrogen compounds with many different oxidized states, the mass transports in sewer become more important.

Since the bottom of sewer was attached by lots of biofilm, wastewater quality in sewer systems changed due to microbial processes during transportation [1]. The transformation of soluble and particulate compounds occurred in bulk water, biofilm and sediment. The variation of quality took place under aerobic as well as anaerobic conditions determined by dissolved oxygen (DO) concen-

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trations. These microbial changes in wastewater substrate as well as in biomass under sewer conditions have been investigated and quantified in many previous studies [2-7].

In order to understand bacterial conversion in biological wastewater treatment processes, many mathematical models were proposed such as Activated Sludge Model (ASM) [8] and applied in our previous works [6, 9-13]. Based on the concept of ASM [8], Hvitved-Jacobsen et al. [14, 15] proposed a mathematical model including six compounds to describe the microbial transformations under variations of aerobic and anaerobic conditions in sewer system. The concept of microbial transformations was based on the biomass growth and substrate utilization used in the activated sludge model. Integration of the aerobic and the anaerobic system for wastewater transformations in sewers made it possible to describe various aerobic/anaerobic conditions. Such changing conditions took place in pressure mains when injecting air or pure oxygen and in gravity sewers with varying slope.

The concentrations of different nitrogen compounds in the domestic wastewater inflow varied with variations of seasons, weather, residents etc. Butler et al. [16] found that the concentrations of ammonia nitrogen ranged from 16 to 52 mg/L and the concentrations of nitrate nitrogen ranged from 0.48 to 1.16 mg/L. Jimenez and Landa [17] presented that the averaged concentrations of total nitrogen and ammonia in the sewage are 33.16 and 29.43 mg/L, respectively. Seidl et al. [18] proposed that the concentrations of ammonia and total Kjeldahl nitrogen (TKN) ranged from 19 to 35 mg/L and 28 to 59 mg/L. High concentrations of nitrate were usually not present in sewer systems, but in some cases, the nitrate was added into the sewer to prevent hydrogen sulfide from producing. Delgado et al. [19] studied the effect of DO in wastewater transformation during transportation. The concentrations of nitrate, nitrite and ammonia varied from 0.5, 0.3 and 39.2 mg/L to 0.4, 0.4

and 38.7 mg/L during transportation. The nitrification and denitrification rates ranged from 0.03 to 0.42 g $(NO_3^--N + NO_2^--N)/m^2hr$ and 0.009 to 0.032 g NO_3^--N/m^2hr.

The interactions between biofilm and nitrogen compounds were very complicated due to various kinds of nitrogen compounds and water flow conditions in sewers. In sewers, two species of autotrophs were capable of oxidizing ammonia to nitrite and subsequently to nitrate under aerobic condition. Then nitrate nitrogen was converted to nitrogen gas by denitrifying bacteria when DO depleted. The biological process by which the ammonia was converted first to nitrite, nitrate and then to nitrogen gas was termed nitrification and denitrification.

The transportation and transformation of pollutants in sewer system were still not fully clear, but well understanding on behaviors of pollutants would result in good operation of treatment plants and maintenance of sewer pipes. So, behaviors of these nitrogen compounds in the sewer systems were an important, worthwhile topic. With this point of view, it is better to understand the transport processes in sewer more clearly, and simulation technique is a good tool to achieve this goal. The objectives of this study were listed as follows. (1) To establish a mathematical model based on the kinetic of ASM to describe the transformation of nitrogen compounds including organic nitrogen, ammonia, and nitrate nitrogen. (2) To explore the effects of varied flow conditions and different initial DO concentration on nitrification and denitrification.

Model development

Hvitved-Jacobsen et al. [14, 15] established a conceptual model that included reaeration and main aerobic and anaerobic microbial processes in water phase and sewer biofilm for wastewater quality changes during transport in sewers, but their model did not include the transformation of nitrogen compounds. In

practice, nitrogen compounds presented in sewer and their kinetics should be considered as shown in Figure 1 [20]. According to ASM, nitrification was assumed to be a one-stage process, directly from ammonia (S_{NH}) to nitrite and nitrate (S_{NO}). Two processes, i.e., autotrophic growth in water phase as well as in biofilm, were considered in this new model. With Monod kinetic, these reactions could be expressed as process rate equation j (aerobic growth of X_{AUT} in bulk water) and process rate equation k (aerobic growth of X_{AUT} in biofilm) in Table 1.

Besides, denitrification was also assumed to be a one-stage process, directly from S_{NO} to nitrogen gas. With Monod kinetic, these reactions could be expressed as process rate equation h (anaerobic growth in bulk water) and process rate equation i (anaerobic growth in biofilm) in Table 1. Additionally, ammonification of soluble organic nitrogen (Process rate equation l in Table 1) and hydrolysis of entrapped organic nitrogen (Process rate equation m in Table 1) were considered in this new model, too. The symbols in these four equation were

explained in Table 2, Table 3, and Table 4. Most of these parameters are directly adopted from previous study [8, 14, 15]. Meanwhile some procedures proposed in our previous works [7, 13] were adopted for calibration.

Model formulation and solution

When the diffusion and convection terms were ignored and the transformation of the nitrogen compounds were considered in the model, combined aerobic/anaerobic processes including transformation of nitrogen are shown in Table 1. The model parameters and compounds used in simulation are shown in Table 2 and Table 3, and the wastewater qualities variation can be expressed as:

$$r_i = \sum_j v_{ij} \rho_j \qquad (1)$$

where r_i = the observed conversion rates of the ith component; v_{ij} = the stoichiometric coefficients of the ith component and jth process rate; ρ_j = the jth process rate.
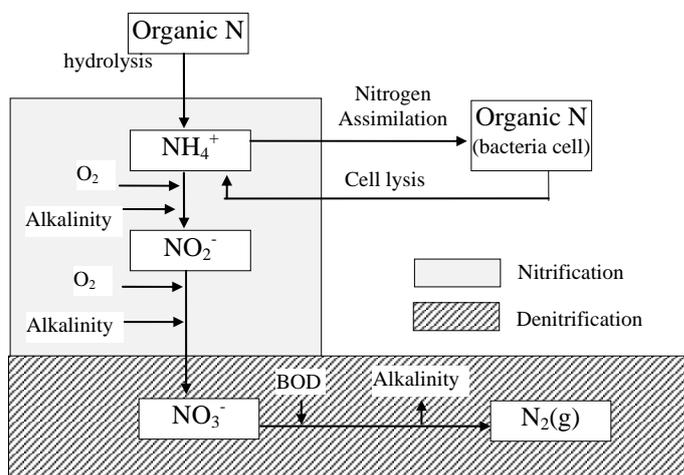

Figure 1. Nitrogen transformations [20].

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Table 1. Integral processes for transformation of wastewater in a sewer.

	S_F	S_A	X_{BW}	X_{S1}	X_{S2}	$-S_O$	X_{AUT}	S_{NO}	S_{NH}	S_{ND}	X_{ND}	Process rate
Aerobic growth in bulk water	$-1/Y_{HW}$	$-1/Y_{HW}$	1			$(1-Y_{HW})/Y_{HW}$			$-i_{XB}$			Eq.a
Aerobic growth in biofilm	$-1/Y_{Hf}$	$-1/Y_{Hf}$	1			$(1-Y_{Hf})/Y_{Hf}$			$-i_{XB}$			Eq.b
Maintenance energy requirement	-1	-1				1						Eq.c
Aerobic hydrolysis, fast	1			-1								Eq.d,n=1
Aerobic hydrolysis, slow	1				-1							Eq.d,n=2
Anaerobic hydrolysis, fast	1			-1								Eq.e,n=1
Anaerobic hydrolysis, slow	1				-1							Eq.e,n=2
Fermentation in bulk water and biofilm	-1	1										Eq.f
Reaeration						-1						Eq.g
Anaerobic growth in bulk water	$-1/Y_{HW}$	$-1/Y_{HW}$	1					$(Y_{HW}-1)/2.86Y_{HW}$	$-i_{XB}$			Eq.h
Anaerobic growth in biofilm	$-1/Y_{Hf}$	$-1/Y_{Hf}$	1					$(Y_{Hf}-1)/2.86Y_{Hf}$	$-i_{XB}$			Eq.i
Aerobic growth of X_{AUT} in bulk water						$(4.57-Y_{AW})/Y_{AW}$	1	$1/Y_{AW}$	$-i_{XB}-Y_{AW}$			Eq.j
Aerobic growth of X_{AUT} in biofilm						$(4.57-Y_{Af})/Y_{Af}$	1	$1/Y_{Af}$	$-i_{XB}-Y_{Af}$			Eq.k
Ammonification of soluble Org-N									1	-1		Eq.l
Hydrolysis of entrapped Org-N										1	-1	Eq.m

a: $\mu_H(S_F+S_A)/(K_S+(S_F+S_A))S_O/(K_O+S_O)X_{BW}\alpha_w^{(T-20)}$

b: $k_{1/2}S_O^{0.5}Y_{Hf}/(1-Y_{Hf})A/V(S_F+S_A)/(K_{Sf}+(S_F+S_A))\alpha_f^{(T-20)}$

c: $q_mS_O/(K_O+S_O)X_{BW}\alpha_w^{(T-20)}$

d: $k_{hn}(X_{Sn}/X_{BW})/(K_{Xn}+X_{Sn}/X_{BW}))S_O/(K_O+S_O)(X_{BW}+\varepsilon X_{Bf}A/V)\alpha_w^{(T-20)}$

e: $\eta_{fe}k_{hn}(X_{Sn}/X_{BW})/(K_{Xn}+X_{Sn}/X_{BW}))S_O/(K_O+S_O)(X_{BW}+\varepsilon X_{Bf}A/V)\alpha_w^{(T-20)}$

f: $q_{fe}S_F/(K_{fe}+S_F)K_O/(K_O+S_O)(X_{BW}+\varepsilon X_{Bf}A/V)\alpha_w^{(T-20)}$

g: $K_La(S_{OS}-S_O)$ where $K_La =0.86(1+0.2F^2)(su)^{3/8}d_m^{-1}\alpha_T^{(T-20)}$

h: $\mu_H\eta_g(S_F+S_A)/(K_S+(S_F+S_A))K_O/(K_O+S_O)S_{NO}/(K_{NO}+S_{NO})X_{BW}\alpha_w^{(T-20)}$

i: $\mu_H\eta_gY_{Hf}/(1-Y_{Hf})A/V(S_F+S_A)/(K_{Sf}+(S_F+S_A))K_O/(K_O+S_O)S_{NO}/(K_{NO}+S_{NO})\alpha_f^{(T-20)}$

j: $\mu_{AUT}S_O/(K_O+S_O)S_{NH}/(K_{NH}+S_{NH})X_{AUT}\alpha_w^{(T-20)}$

k: $k_{1/2}S_O^{0.5}Y_{Af}/(1-Y_{Af})A/V S_O/(K_O+S_O)S_{NH}/(K_{NH}+S_{NH})\alpha_f^{(T-20)}$

l: $k_aS_{ND}(X_{BW}+X_{Bf})$

m: k_7X_{ND}

Table 2. Model parameters used in the sewer process model outlined in Table 1.

Symbol and definition		value	Unit
μ_H	maximum specific growth rate for heterotrophic biomass	7	d^{-1}
μ_{AUT}	maximum specific growth rate for autotrophic biomass	0.8	d^{-1}
Y_{HW}	yield constant for suspended heterotrophic biomass	0.55	gCOD/ gCOD
Y_{Hf}	yield constant for biofilm heterotrophic biomass	0.55	gCOD/ gCOD
Y_{AW}	yield constant for suspended autotrophic biomass	0.24	gCOD/ gN
Y_{Af}	yield constant for biofilm autotrophic biomass	0.24	gCOD/ gN
K_S	saturation constant for readily biodegradable substrate	1.0	$gCOD/m^3$
K_O	saturation constant for DO	0.5	gO_2/m^3
K_{NH}	ammonia half-saturation constant for autotrophic biomass	1.0	$gNH_3\text{-}N/m^3$
K_{NO}	nitrate half-saturation constant for denitrifying heterotrophic biomass	0.5	$gNO_3\text{-}N/m^3$
K_{Sf}	saturation constant for readily biodegaradable substrate	1.0	$gCOD/m^3$
α_W	temperature coefficient in the water phase	1.07	-
α_f	temperature coefficient in the biofilm	1.03	-
q_m	maintenance energy requirement rate constant	1.0	d^{-1}
$k_{1/2}$	1/2 order rate constant	2.5	$gO_2^{0.5}m^{-0.5}d^{-1}$
ε	efficiency constant for the biomass	-	-
η_g	correction factor for μ_H under anoxic conditions	0.8	-
k_{h1}	hydrolysis rate constant, fraction 1 (fast)	4.0	d^{-1}
k_{h2}	hydrolysis rate constant, fraction 2 (slow)	1.0	d^{-1}
K_{X1}	saturation constant for hydrolysis, fraction1	0.5	gCOD/ gCOD
K_{X2}	saturation constant for hydrolysis, fraction2	0.2	gCOD/ gCOD
η_{fe}	anaerobic hydrolysis reduction factor	0.14	-
q_{fe}	maximum rate for fermentation	3	d^{-1}
K_{fe}	saturation constant for fermentation	20	gCOD/ gCOD

Table 2. Model parameters used in the sewer process model outlined in Table 1. (continued)

	Symbol and definition	value	Unit
k_a	ammonification rate	0.08	$m^3COD(g\cdot d)^{-1}$
k_7	hydrolysis rate of entrapped Org-N	2.0	d^{-1}
i_{XB}	mass N/mass COD in biomass	0.086	gN/gCOD

Table 3. The concentrations of different components in wastewater for model simulation.

	Component	Characteristic value, wastewater	For simulation	Unit
X_{BW}	heterotrophic active biomass in water phase	20-100	5	$gCOD/m^3$
X_{Bf}	heterotrophic active biomass in biofilm	-	10	$gCOD/m^3$
X_{S1}	hydrolysible substrate, fast biodegradable	50-100	42.5	$gCOD/m^3$
X_{S2}	hydrolysible substrate,slowly biodegradabe	300-450	85	$gCOD/m^3$
S_S	readily biodegradable substrate	0-50	42.5^{++}	$gCOD/m^3$
S_O	dissolved oxygen	0-4	3	gO_2/m^3
X_{AUT}	autotrophic active biomass	-	0.2	$gCOD/m^3$
S_{NO}	nitrate and nitrite nitrogen		3	$g\ NO_2^-+NO_3^-/m^3$
S_{NH}	NH_4^+ and NH_3 nitrogen	20-50	20	gNH_3/m^3
S_{ND}	soluble biodegradable organic nitrogen		7	$gOrgN/m^3$
X_{ND}	particulate biodegradable organic nitrogen		10	$gOrgN/m^3$

*include very slowly biodegradable and inert organic matter.   **$S_S=S_F+S_A$; S_F=25.5 gCOD/ and S_A=17gCOD/m^3

Table 4. Reaeration and flow characteristics used in the sewer process model outlined in Table 1.[4]

	Symbol and definition	value	Unit
K_{La}	oxygen transfer coefficient		d^{-1}
T	temperature	23.4	^oC
S_{OS}	dissolved oxygen saturation concentration at T^oC	8.4	gO_2/m^3
F	Froude number $=u/(g\ d_m)^{0.5}$	varied	-
u	mean flow velocity	varied	m/s
g	gravity acceleration	9.81	m/s^2
s	slope	0.01	m/m
d_m	hydraulic mean depth	0.15	m
α_r	temperature coefficient for reaeration	1.024	
A/V	ratio of area to volume	8	m^2/m^3

The equations that described the transport and transformation of wastewater quality in this model formed an ordinary differential equation (ODE) system. The set of eleven equations in this model was solved simultaneously using fourth-order Runge-Kutta numerical method [21]. The flow characteristics and reaeration used in the model are shown in Table 4.

Results and discussion

This model was applied on a 10 km long, 0.15 m diameter, 0.3 m depth of sewage flow and 0.1 m depth of sediments sewer system. Since the DO concentrations that affected the biological activities in the sewer system varied with the variations of flow velocities, simulations were divided into two parts: (1) first, the flow velocity was fixed, the initial DO concentrations were viewed as variables, (2) second, the initial DO concentration was fixed, and flow velocities were taken as variables. When the flow velocity was fixed, the

initial DO concentrations were assumed to be 0.0, 3.0, 6.0 mg/L, respectively. When the DO concentration was fixed, the flow velocities were assumed to be 0.6, 0.9, 1.2 m/s, respectively. The simulation results are shown in Figure 2 and Figure 3, respectively.
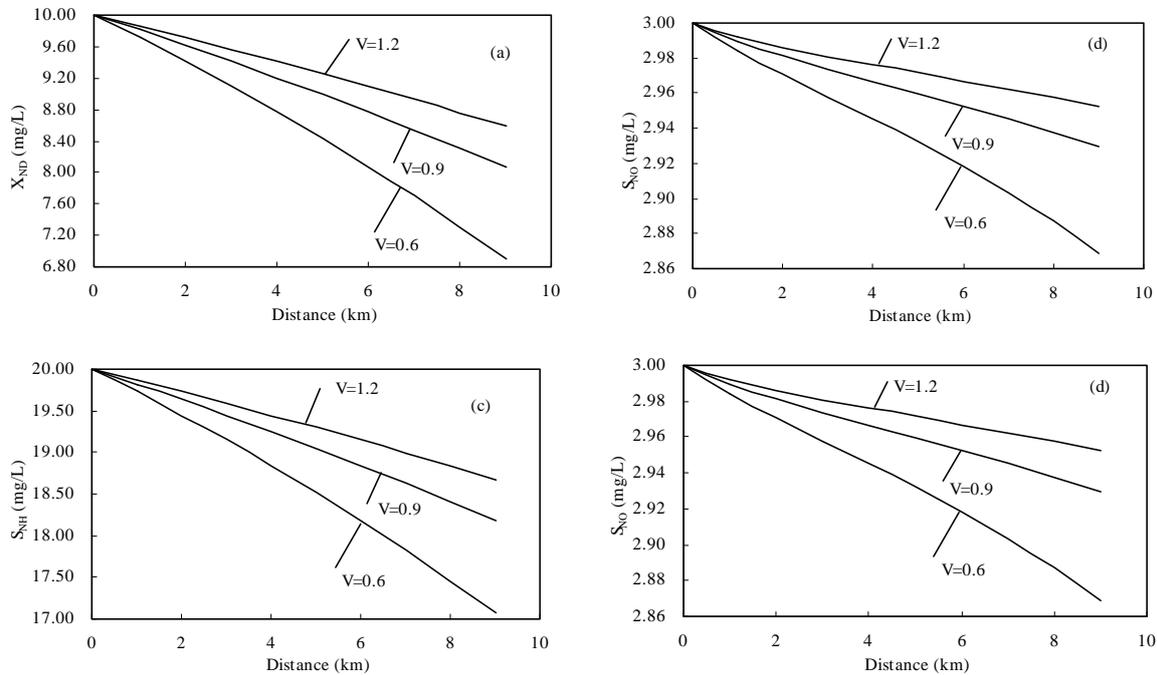

Figure 2. The variations of (a) X_{ND}, (b) S_{ND}, (c) S_{NH}, and (d) S_{NO} with flowing distances at different DO level (when flow velocity = 0.9 m/s).

According to simulation, the variations of particulate biodegradable organic nitrogen, X_{ND}, were identical in each simulation, because only the first order hydrolysis kinetic reaction affected the concentrations and the reaction time was same at consistent flow velocity (Figure 2 (a)). Figure 2 (b) shows the variations of soluble biodegradable organic nitrogen, S_{ND}, with the flowing distance at different DO levels. Although the process rate was affected by the soluble biodegradable organic nitrogen concentrations and heterotrophic biomass, the concentration variations were almost the same due to the balance between ammonification of soluble organic nitrogen (process rate equation l in Table 1) and hydrolysis of entrapped organic nitrogen (process rate equation m). When DO increased from 0.0 mg/L to 6.0 mg/L, S_{ND} only increased by 0.35 %. The variations of the ammonium and ammonia nitrogen concentrations, S_{NH}, are shown in Figure 2 (c). Their concentrations declined with flowing distance, and were higher when DO at lower level. It resulted from the net reaction including aerobic growth of X_H in bulk water (process rate equation a), aerobic growth of X_H in biofilm (process rate equation b), anaerobic growth of X_H in bulk water (process rate equation h), anaerobic growth of X_H in biofilm (process rate equation i), aerobic growth of X_{AUT} in bulk water (process rate equation j), aerobic growth of X_{AUT} in biofilm (process rate equation k) and ammonification of soluble organic nitrogen (process rate equation l). When DO increasing from 0.0 mg/L to 6.0 mg/L, S_{NH} only decreased by 0.12 %. According to Figure 2 (d), the nitrite and nitrate nitrogen concentrations in the water column, S_{NO}, decreased with flowing distance, and were lower

when DO at a lower level. S_{NO} decreased by 2.22 % as DO decreasing from 6.0 mg/L to 0.0 mg/L. The variations represented that the denitrification rate was faster that of nitrification due to denitrification was enhanced at low DO level.

When the initial DO concentrations were fixed at 3.0 mg/L, the particulate biodegradable organic nitrogen, X_{ND}, declined with the flowing distance, and was lower at low flow velocity, as shown in Figure 3 (a). Although only the first order hydrolysis kinetic reaction affected the concentrations and the flowing distances were the same in each simulation, the reaction time was different due to different flow velocities. When flow velocity was lower, the reaction time was longer. It resulted in greater reduction of X_{ND} under slow-flowing conditions. When flowing velocity increased from 0.6 m/s to 1.2 m/s, X_{ND} concentrations increased by 24.40 %. Figure 3 (b) shows the variations of soluble biodegradable organic nitrogen, S_{ND}, with the flowing distance at different flow condition.

S_{ND} decreased with flowing distance, too. When flowing velocity increased from 0.6 m/s to 1.2 m/s, S_{ND} concentrations increased by 14.52 %. The variations of the ammonium and ammonia nitrogen concentrations, S_{NH}, are shown in Figure 3 (c). Their concentrations declined with flowing distance and were higher when flow velocity was greater. When flowing velocity increased from 0.6 m/s to 1.2 m/s, S_{NH} concentrations increased by 9.43 %. According to Figure 2 (d), the nitrite and nitrate nitrogen concentrations in the water column, S_{NO}, decreased with the flowing distance. When flowing velocity increased from 0.6 m/s to 1.2 m/s, S_{NO} concentration increased by 2.91 %.

According to simulation, when DO varied and flow velocities were fixed, all nitrogen compounds varied slightly excepting S_{NO}. When initial DO was fixed, the different reaction time due to different flow velocity affected the nitrogen transformation significantly.
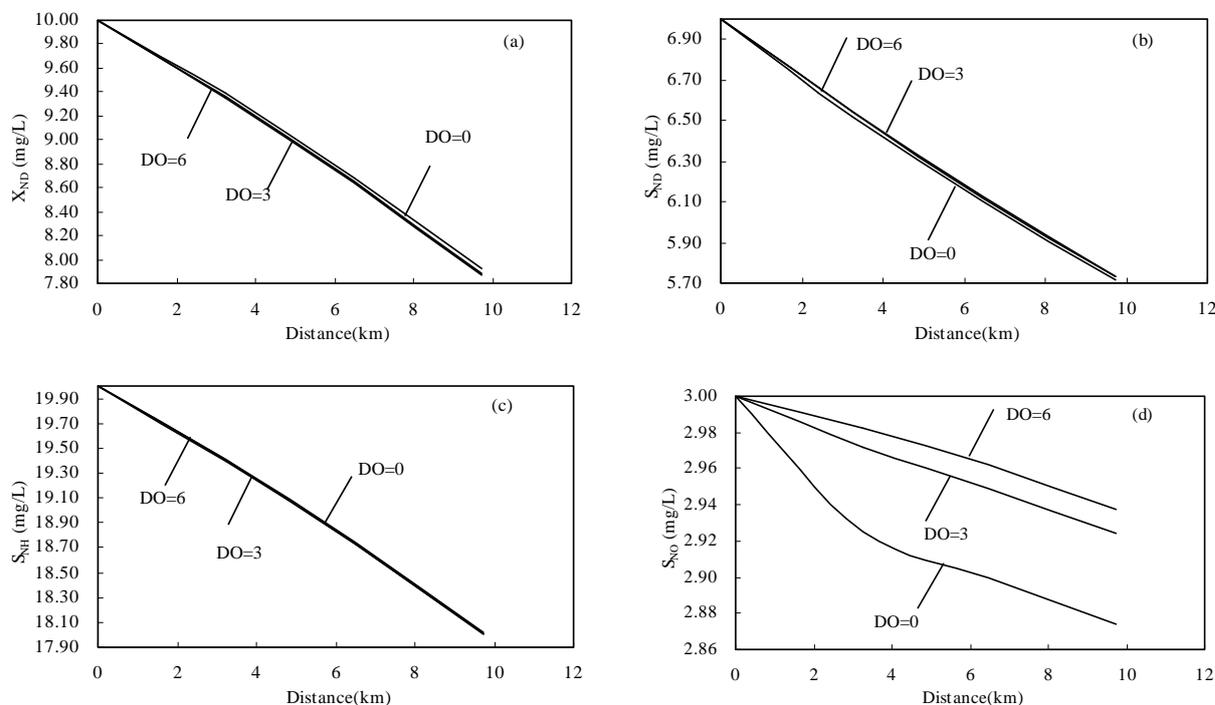

Figure 3. The variations of (a) X_{ND}, (b) S_{ND}, (c) S_{NH}, and (d) S_{NO} with flowing distances at different flow conditions (when initial DO concentration = 3.0 mg/L).

Conclusions

The concentrations of nitrogen compounds including X_{ND}, S_{ND}, S_{NH}, and S_{NO} in the bulk water changed due to physical, chemical and biological reactions. A model including transformation of nitrogen compounds was established to describe the behaviors of these four nitrogen compounds in the sewer systems. The results showed that these four compounds were transformed during the transportation process.

According to simulation, when DO varied and flow velocities were fixed, all nitrogen compounds varied slightly excepting S_{NO}. When initial DO was fixed, the different reaction time due to different flow velocity affected the nitrogen transformation significantly. Removal of nitrogen compounds was better when reaction time was greater. To design a self-purification sewer system or enhance the removal of pollutants in sewer can remove pollutants from sewage before flowing into wastewater treatment plant, and can avoid the accumulation of pollutants in the sewer system.

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