

Transport in Porous Media (2005) 60:319–337
DOI 10.1007/s11242-004-6325-z

© Springer 2005

Modeling Floodplain Filtration for the Improvement of River Water Quality

JONG-BAE CHUNG¹, SEUNG-HYUN KIM^{2,*}, BYEONG-RYONG JEONG³, YOUNG-DEUK LEE¹ and SHIV O. PRASHER⁴

¹*Department of Agricultural Chemistry, Daegu University, Gyeongsan, South Korea 712-714*

²*Department of Environmental Engineering, Yeungnam University, Gyeongsan, South Korea 712-749*

³*Department of Agronomy, Daegu University, Gyeongsan, South Korea 712-714*

⁴*Department of Agricultural and Biosystems Engineering, McGill University, 21,111 Lakeshore, Ste-Anne-de-Bellevue QC, Canada H9X 3V9*

(Received: 12 May 2004; accepted in final form: 14 November 2004)

Abstract. A mathematical model was developed to describe a treatment method of floodplain filtration for the improvement of river water quality. The process consists of spraying poor quality river water onto the river floodplains and thus allowing soil filtration to treat water before it gets back again into the main river stream. This technique can be readily employed in Korea because it exploits the characteristics of the climate and rivers in the country, as described in an experimental study of Chung *et al.* (2004). The model was analyzed by numerical methods and validated by comparing the simulated values with experimental data. A scenario analysis of the model was also performed in order to have a better understanding of the floodplain filtration process. Our results show that the model was able to predict the reduction in organic matter and NO_3^- in river water through the floodplain filtration. Furthermore, it was found that only a few decimeters of top soil profile were enough to degrade most of the organic matter under wider operational conditions than those reported in the literature. Also, it was found that significant infiltration of atmospheric oxygen took place near the soil surface. The N_2O emission and the NO_3^- leaching increased with the increase in the influent NO_3^- concentration. However, the N_2O emission due to floodplain filtration was not expected to exceed $0.1 \text{ mL/m}^2\text{-day}$.

Key words: floodplain filtration, organic matter removal, denitrification, mathematical modeling, competitive Michaelis–Menten model.

1. Introduction

The rivers in Korea are characterized by their wide floodplains which are prepared for the flooding due to the monsoons and typhoons. The flooding

*Author for correspondence: Tel.: +82-53-810-2548; Fax: +82-53-810-4624;
e-mail: kimsh@yumail.ac.kr

occurs several times in a year and it lasts only for less than a week. The floodplains remain uncultivated and weedy for the most part of the year. So, the floodplains can potentially be exploited for the treatment of the contaminated river water.

The treatability of contaminated river waters in Korea was investigated experimentally by Chung *et al.* (2004) using a floodplain soil. They performed a soil column experiment simulating the floodplain filtration (Figure 1) and concluded that floodplain filtration could successfully remove both organic matter and NO_3^- simultaneously from the contaminated river waters. In their experiment, the column soil was kept unsaturated and the aerobic conditions in the column were controlled by managing the water spray rate. A denitrifying zone developed under the aerobic zone. In another experiment by Kim *et al.* (2003a), it was found that the water spray rate could be raised successfully up to $210 \text{ L/m}^2\text{-day}$. They obtained similar results to those reported by Chung *et al.* (2004) without any clogging problem. In the study by Kim *et al.* (2003a), contrary to the study of Chung *et al.* (2004), the soil surface was covered with sod and the secondary effluent of a municipal wastewater treatment plant was used as the poor quality water. Before this method is adopted for the treatment of poor quality water in a practical scale, there is a need to make further investigations or simulate the floodplain filtration process under a wide range of conditions. Also, we need to estimate the N_2O emissions that can be caused by this technique.

For describing the behavior of nitrogen oxides during the floodplain filtration process the competitive Michaelis–Menten model could be a good candidate since it can take into account the concentration of each nitrogen oxide separately (Cho and Mills, 1979; Cho *et al.*, 1997a,b). However, the electron affinity coefficients, needed by this model, are not easy

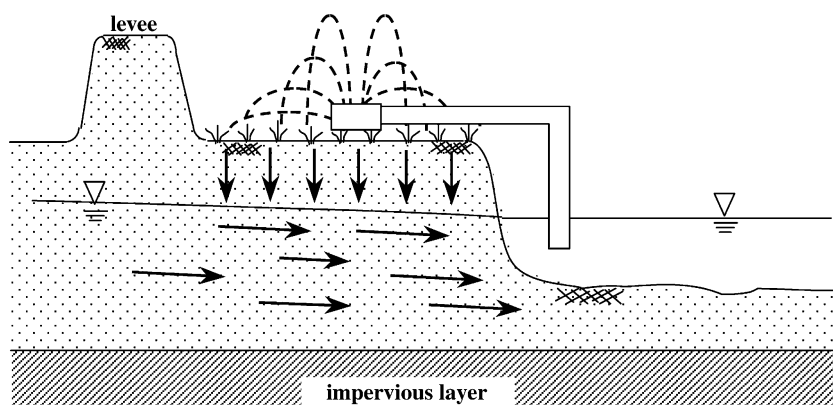


Figure 1. Schematization of the floodplain filtration technique.

to determine (Dendooven *et al.*, 1994). Recently, an experimental study was reported by Kim *et al.* (2003b) to determine these coefficients, along with other biodegradation kinetic parameters, using a typical Korean river water under both aerobic and denitrifying conditions.

In the present study, a mathematical model was developed to describe the floodplain filtration process. The model focused on the simultaneous removal of organic matter and NO_3^- . The model input parameters were obtained from published literature. The model was solved numerically and its predictions were compared with experimental results, reported previously in published literature. It was also run for different operating conditions in order to obtain a better understanding of the floodplain filtration process.

2. Model Development

Mathematical models describe the behavior of organic matter, electron acceptors, and biomass in the unsaturated floodplain soil based on their mass balances. Since floodplains are very wide compared to the vertical depths of their unsaturated zone, the domain of this study is regarded as vertical and one-dimensional. Organic matter is assumed to migrate in the unsaturated zone of a floodplain by advection experiencing biodegradation and to be released from the solid matrix, and its behavior may be described as (de Marsily, 1986)

$$R \frac{\partial S}{\partial t} = - \frac{\partial}{\partial z} (v_w \theta_w S) - \frac{1}{Y} \frac{\mu_{\max} \eta S}{K_S + \eta S} X \theta_w + 1.42 \times 0.9 \times k_d X \theta_w + k_S S_S \quad (1)$$

where S is the aqueous phase concentration of organic matter measured in mg-BCOD/L (where BCOD signifies Biodegradable Chemical Oxygen Demand), S_S denotes the organic content of the solid matrix (mg-organic matter/kg-soil), R means the retardation factor (—) due to the adsorption and desorption of the organic matter on the solid surface (Domenico and Schwartz, 1990), v_w stands for the linear ground water velocity (cm/s), and θ_w represents the volumetric water content (—). The parameter η expresses the average substrate availability to the microbes, i.e., the effectiveness factor (—). The parameter Y is the yield coefficient (mg-MLVSS/mg-BCOD), μ_{\max} means the maximum specific growth rate (1/s), K_S represents the half saturation constant (mg-BCOD/L), X signifies the aqueous phase biomass concentration (mg-MLVSS/L), and k_d denotes the microbial decay rate coefficient (1/s). The k_S represents the release rate coefficient of organic matter from the solid matrix (kg-soil \times mg-BCOD/L-total soil volume-mg-organic matter-s).

In this study the model is derived for the transient situation, however, it is to be applied to represent a continuously running operation, i.e., steady

state. The retardation factor does not affect the spatial distribution of the organic matter concentration at steady state (Kim and Corapcioglu, 1996). The organic matter in river waters is somewhat recalcitrant; however, if captured in soil it can have enough time to be biodegraded due to the extended contact time with the biomass in the soil, which can increase biodegradability of the organic matter. On this basis the BCOD in this study was regarded as the difference between the COD at the soil surface and at some depth in the soil. The Monod model, as a modified form, was employed for the microbial growth. The Monod parameters, Y , μ_{\max} , K_S , and k_d , have different values under different oxidation conditions (Doussan *et al.*, 1997), which is taken into account in this study by categorizing them as the aerobic and the denitrifying zones. Furthermore, it was assumed that the consumption rate of the organic matter was not related to the DO (Dissolved Oxygen) level in the aerobic reaction and also not related to the kind and concentration of each nitrogen oxide in denitrification (Cho, 1982; Dendooven *et al.*, 1994).

Ninety percent of the dead biomass was assumed to return to the substrate again (Lensing *et al.*, 1994), and 160 mg of DO was assumed to be required in the decomposition of 113 mg of the biomass yielding 1.42 as the ratio (Ramalho, 1983). Some time is required in the decomposition of the dead biomass to serve as the substrate; however, this lag was ignored since the growth and the decay rates of the biomass are identical at steady state. In the case of very low substrate concentration in the aqueous phase, the organic matter released from the solid matrix can be a major source of the substrate utilized by the biomass (Kinzelbach *et al.*, 1991). The release was described as a first order reaction (Lensing *et al.*, 1994). During the release, the organic content of the solid matrix may decrease. However, it was assumed to be constant since the recalcitrant organic matter can be supplied from the upstream, adsorb on the surface, and be released again as a biodegradable form in some time. In addition, the organic matter from the solid matrix and the dead biomass were assumed to have the same kinetic characteristics of the biodegradation reaction as the organic matter in the river water.

The soil bacteria involved in floodplain filtration are expected to grow mostly in the form of biofilm (Rittmann, 1993), of which only a small part is occupied by the net biomass (Characklis *et al.*, 1991), and the concentration ranges within 10,000–100,000 mg/L and does not vary significantly with time and space (Rittmann, 1993). In this sense, the microbial growth signifies an increase in the volume of the biofilm. However, since the purpose of this study is not to understand the clogging of the porous medium, the microbial growth is assumed to take place over the entire aqueous phase (Baveye and Valocchi, 1989; Clement *et al.*, 1996) to be described as;

$$\frac{dX}{dt} = \frac{\mu_{\max} \eta S}{K_S + \eta S} X - k_d X \quad (2)$$

Although the microbes are assumed for computational simplicity to be distributed over the entire aqueous phase in Equation (1), they are still in the form of biofilm, and the microbes cannot utilize the high concentration of substrate in the aqueous phase. The average substrate availability was assumed in this study to be the ratio of the net water content to the sum of the net water content and the volume of the biofilm.

In floodplain filtration, with the presence of oxygen, the decomposition of organic matter is accompanied by the consumption of oxygen. The oxygen is supplied from the sprayed river water as dissolved oxygen and from the atmosphere by diffusion through the unsaturated soil, and its transport can be expressed as

$$\begin{aligned} \frac{\partial}{\partial t} (\theta_w C_{O_2} + \theta_a K_{O_2} C_{O_2}) = & \frac{\partial}{\partial z} \left(D_{a,O_2} K_{O_2} \frac{\partial C_{O_2}}{\partial z} \right) - \\ & - \frac{\partial}{\partial z} (\theta_w v_w C_{O_2}) - \theta_w \frac{a_{O_2}}{Y} \frac{\mu_{\max} \eta S}{K_S + \eta S} X \end{aligned} \quad (3)$$

where θ_a is the volumetric air content (—) which can be obtained from $\theta_a = n - \theta_w$, n means the porosity of the clean medium (—), C_{O_2} denotes the DO concentration (mg-O₂/L of water), $K_{O_2} = C_{O_2}^g / C_{O_2}$ represents the equilibrium distribution coefficient of oxygen between air and water phases (—), $C_{O_2}^g$ expresses the gas phase oxygen concentration (mg-O₂/L of gas), D_{a,O_2} stands for the effective diffusivity through the soil (cm²/s), and a_{O_2} indicates the oxygen consumption rate of the substrate during biodegradation (mg-O₂ Consumed/mg-BCOD Decomposed). The oxygen consumption owing to the microbial decay is not considered here since the microbial decay has already been regarded as the production of the organic substrate. The oxygen was assumed to migrate through the soil gas by only diffusion and through the soil water by only advection, and the equilibrium concentration was assumed between the two phases (Cho *et al.*, 1997b).

In floodplain filtration, the oxygen functions as the electron acceptor near the soil surface, however, its concentration becomes lower with the distance from the surface until it is completely exhausted. With no oxygen, the nitrogen oxides, including NO₃[−], NO₂[−], and N₂O, function as the acceptor, and after their depletion manganese and iron become the acceptor (Patrick and Jugsujinda, 1992). N₂ and N₂O gases are the products of denitrification, and N₂O has the global warming effect if emitted to the atmosphere, which may be a defect of the floodplain filtration technique and makes it important to quantify its emission. In this study, as the competitive Michaelis–Menten model of Cho *et al.* (1997a) was employed it

is possible to investigate the behavior of N_2O along with that of other nitrogen oxides. The behavior of NO_3^- in the floodplain soil employing the model with the assumption of no dispersion can be described as

$$\theta_w \frac{\partial C_{\text{N5}}}{\partial t} = -\frac{\partial}{\partial z} (\theta_w v_w C_{\text{N5}}) - \theta_w \frac{Q_{\text{N5}} C_{\text{N5}}}{2Q_{\text{N5}} C_{\text{N5}} + 4Q_{\text{N3}} C_{\text{N3}}^2 + 2Q_{\text{N1}} C_{\text{N1}}} R_n \quad (4)$$

where C_{N5} , C_{N3} , and C_{N1} are the aqueous phase concentrations of NO_3^- , NO_2^- , and N_2O , respectively, and have the same unit of mmol/L, and Q_{N5} , Q_{N3} , and Q_{N1} mean, respectively, the electron affinity coefficients of NO_3^- , NO_2^- , and N_2O , of which the units are L/mmol, L^2/mmol^2 , and L/mmol, respectively. R_n denotes the electron production rate (mmol/L-s) in the soil water phase (Cho *et al.*, 1997a; Cho and Mills, 1979). Since denitrification is a microbial reaction, the electron production rate is related to the kind and concentration of the substrate (Burford and Bremner, 1975). In the case of high substrate concentration owing to the introduction of contaminated river water, the modified Monod model can be employed to express the rate as

$$R_n = \frac{a_N}{Y} \frac{\mu_{\max} \eta S}{K_s + \eta S} X \quad (5)$$

where a_N is the ratio of the electron production to the substrate consumption in the denitrification process and becomes 0.125 (mmol-Electron Produced/mg-BCOD Removed) with the assumption of no substrate uptake for the biomass formation.

The nitrite behavior in the floodplain soil, following the assumption used for NO_3^- , can be expressed as

$$\theta_w \frac{\partial C_{\text{N3}}}{\partial t} = -\frac{\partial}{\partial z} (\theta_w v_w C_{\text{N3}}) + \theta_w \frac{Q_{\text{N5}} C_{\text{N5}} - 2Q_{\text{N3}} C_{\text{N3}}^2}{2Q_{\text{N5}} C_{\text{N5}} + 4Q_{\text{N3}} C_{\text{N3}}^2 + 2Q_{\text{N1}} C_{\text{N1}}} R_n \quad (6)$$

In the modeling of the transport of N_2O and N_2 gases in the soil, the transport through both the soil water and the soil gas should be considered at the same time. With the same assumptions made in the oxygen transport equation, the migration of the N_2O and N_2 can be described as

$$\begin{aligned} \frac{\partial}{\partial t} (\theta_w C_{\text{N1}} + \theta_a K_{\text{N1}} C_{\text{N1}}) = & \frac{\partial}{\partial z} \left(D_{a,\text{N1}} K_{\text{N1}} \frac{\partial C_{\text{N1}}}{\partial z} \right) - \frac{\partial}{\partial z} (\theta_w v_w C_{\text{N1}}) + \\ & + \theta_w \frac{Q_{\text{N3}} C_{\text{N3}}^2 - Q_{\text{N1}} C_{\text{N1}}}{2Q_{\text{N5}} C_{\text{N5}} + 4Q_{\text{N3}} C_{\text{N3}}^2 + 2Q_{\text{N1}} C_{\text{N1}}} R_n \end{aligned} \quad (7)$$

and

$$\begin{aligned} \frac{\partial}{\partial t} (\theta_w C_{N0} + \theta_a K_{N0} C_{N0}) = & \frac{\partial}{\partial z} \left(D_{a,N0} K_{N0} \frac{\partial C_{N0}}{\partial z} \right) - \frac{\partial}{\partial z} (\theta_w v_w C_{N0}) + \\ & + \theta_w \frac{Q_{N1} C_{N1}}{2Q_{N5} C_{N5} + 4Q_{N3} C_{N3}^2 + 2Q_{N5} C_{N1}} R_n \quad (8) \end{aligned}$$

respectively (Cho *et al.*, 1997b), where C_{N0} means the aqueous phase concentration of N_2 (mmol/L), K_{N1} and K_{N0} are the equilibrium distribution coefficients of N_2O and N_2 , respectively, between gas and aqueous phases (–). $D_{a,N1}$ and $D_{a,N0}$ represent the effective diffusivities of N_2O and N_2 gases through the soil, respectively. It was further assumed that the aerobic and denitrifying zones could be distinguished along the soil depth and that other factors such as pH (Burford and Bremner, 1975; Stanford *et al.*, 1975) had only a negligible effect on the denitrification in soil. In this study the temperature is fixed, however, it can be extended to match the temperature variations as in other studies (Cho *et al.*, 1979).

3. Determination of Model Parameters

The model parameters employed in the model validation were obtained mostly from the literature. The Monod parameters for both aerobic and denitrifying conditions, the electron affinity coefficients of the nitrogen oxides, and the a_{O_2} were obtained from the study of Kim *et al.* (2003b) where they performed the experiment employing the same conditions as those in Chung *et al.* (2004). The volume of the biofilm for the calculation of the average substrate availability was obtained by dividing the biomass by the bacterial concentration in the biofilm. The bacterial concentration was estimated as 7500–15,000 mg/L in the study of Sung *et al.* (2003), where the same raw water was used as in the studies of Kim *et al.* (2003b) and Chung *et al.* (2004).

The θ_w in Equation (1) was obtained through the analysis of the Richards equation (Guymon, 1994) using the experimental conditions employed in Chung *et al.* (2004). The soil moisture retention curve was obtained through the pressure plate experiment and was assumed to follow the van Genuchten's model (Guymon, 1994). In this process the van Genuchten's model was fitted to the result of the pressure plate experiment to yield best fitting parameters of the model. The procedure of this method applied to another soil is shown in Kim *et al.* (2001). The saturated hydraulic conductivity was determined using a constant head permeameter, of which the diameter and length were 15 and 30 cm, respectively. The permeameter was filled with soil as in the experiment of Chung *et al.* (2004) and was exposed to a constant head difference of 30 cm, resulting in upward water flow. Then the flow rate was measured to calculate

the hydraulic conductivity using the Darcy's law (Bear, 1972). The unsaturated hydraulic conductivity of the soil was assumed to be constant throughout the soil depth for simplification in spite of the difference in biomass with soil depth. Moreover, in order to further simplify the computation, the change in the soil moisture retention characteristics, which may be caused by the microbial growth, was neglected. The equilibrium distribution coefficients K_{O_2} , K_{N1} , K_{N0} were obtained from the literature (Dean, 1999). The effective diffusivity of oxygen in soil was determined according to Millington and Shearer (1971) using

$$D_{a,O_2} = D_{O_2} \left(1 - \frac{\theta_w}{n}\right)^2 (n - \theta_w)^{2x} \quad (9)$$

where D_{O_2} means the molecular diffusivity (cm^2/s) in the atmosphere at 20°C , and x is a parameter which is obtainable (Collin and Rasmuson, 1988; Quyang and Boersma, 1992) from the solution to the equation of

$$(\theta_a)^{2x} + (1 - \theta_a)^x = 1 \quad (10)$$

4. Summary of the Experimental Results Used in the Model Validation

Some characteristics of the stream water, which was used as the raw water in the experiments of Kim *et al.* (2003b) and Chung *et al.* (2004), are listed in Table I. The floodplain soil used in the study of Chung *et al.* (2004) has the characteristics listed in Table II. The van Genuchten parameters for the unsaturated soil were determined to be $\alpha = 0.061$ and $N = 1.63$. The saturated hydraulic conductivity of the soil was $9.68 \times 10^{-4} \text{ cm/s}$. The parameters of the biodegradation reaction kinetics and the electron affinity coefficients are summarized in Table III. The average difference between the COD of the influent and at the soil depth of 90 cm of the lysimeters in Chung *et al.* (2004) was regarded as the BCOD of the raw water. About 75% of the COD of the river water was the BCOD, which shows similar result to the report of Kim *et al.* (2003b).

Table I. Some chemical characteristics of the raw water used in the experiments of Kim *et al.* (2003b) and Chung *et al.* (2004)

	pH	DO (mg/L)	Eh (mV)	COD (mg/L)	BCOD (mg/L)	NO ₃ -N (mg/L)	NO ₂ -N (mg/L)	NH ₄ -N (mg/L)
Average	7.6	5.2	294.8	17.1	11.9	2.03	0.12	0.68
Range	7.3–8.0	4.8–5.4	287.4–301.7	16.5–18.1	10.3–16.4	1.89–3.02	0.09–0.18	0.59–0.81

Table II. Physical and chemical characteristics of the soil employed in the study of Chung *et al.* (2004)

pH	Particle density (g/cm ³)	Organic matter (%)	CEC [†] (cmol/kg ¹)	Particle separate		
				Sand (%)	Silt (%)	Clay (%)
5.39	2.56	0.45	2.03	83.3	16.4	0.3

[†]Cation exchange capacity.

Table III. Summary of the bioreaction parameters and the electron affinity coefficients from Kim *et al.* (2003b)

Bioreaction parameters	Y	μ	k_d	K_S	a_{O_2}
	Yield coeff.	Maximum specific growth rate	Cell decay rate coeff.	Monod half coeff.	Oxygen consumption rate
	mg-MLVSS/ mg-BCOD	day ⁻¹	day ⁻¹	mg-BCOD L ⁻¹	mg-O ₂ consumed/ mg-COD removed
Aerobic	0.59	5.32	0.042	35.3	0.29–0.68
Anoxic	0.413	1.51	0.037	29.43	–
Electron affinity coefficients	Q_{N_5} L/mmol of NO ₃ ⁻		Q_{N_3} L ² /mmol ² of NO ₂ ⁻		Q_{N_1} L/mmol of N ₂ O
	1		306		131

5. Numerical Algorithms

In order to understand the behavior of the water quality parameters in the floodplain, the Equations (1)–(8) were solved simultaneously. Although the system in this study was limited only to the steady state, the equations were solved in terms of time as well as space. This computational manipulation was introduced because the variables in the equations interfere with each other resulting in difficulties analyzing even in the case of steady state, and it was also possible because a transient solution can reach steady state with enough time. In the analysis of the equations, numerical methods were employed.

In the solution to the Richards equation, the fully implicit scheme and the predictor–corrector method were employed to overcome the strong nonlinearity, and the constant application rate boundary condition at the upper end and the constant water pressure boundary condition at the lower end were applied. The same algorithm was applied as the one that was already verified by comparing with Hills *et al.* (1989) in other study Kim *et al.*, 1998. The solution is not shown in this study, however, the water content remained constant throughout the soil depth except at some lower part (Kim *et al.*, 2001).

Equations including advection terms were solved by the MOC (Method of Characteristics) in order to minimize the numerical dispersion (Molz *et al.*, 1986). The dispersion terms were analyzed by the finite difference method (fully implicit centered difference scheme) while the reaction terms were solved by the Runge–Kutta method of fourth order. The solutions with no reaction terms were compared with analytical ones obtained from van Genuchten and Alves (1982), and they were in good agreement (Sung *et al.*, 2003; Kim *et al.*, 2000). The sequential procedure was applied in the solution of the system of equations. For all the second order equations the constant concentration condition was applied to the upper boundary, while no concentration gradient was assumed at the lower ends. Equation (10) was solved by a trial-and-error method.

6. Model Validation and Scenario Analysis

The model predictions were compared for validation to the results of the lysimeter experiment reported by Chung *et al.* (2004). Model predictions of the concentrations of the BCOD, DO, nitrogen oxides, and nitrogen are shown in Figure 2 compared with the experimental data of the lysimeter with the application rate of 68.0 L/m²-day. At this rate, the denitrification process took place actively as reported in Chung *et al.* (2004). The parameters employed in the prediction are listed in Table IV. The parameters in Table III, which was from the study of Kim *et al.* (2003b), were changed into the values shown in Table IV through model calibration. This change was required due presumably to the discrepancy between the reactor and the soil environments. From the change it can be seen that the microbial growth and decay rates are much lower in column soil than in the bio-reactor contrary to the electron affinity coefficients of nitrite and nitrous oxide. In order to obtain the prediction curves, boundary and initial conditions representative of the actual situation were applied in the analysis of the governing equations. The organic release from the soil was neglected.

Figure 2 shows that the concentration predictions are in fairly good agreement with the experimental result of Chung *et al.* (2004). It can also be seen that the BCOD and DO concentrations decrease rapidly near the

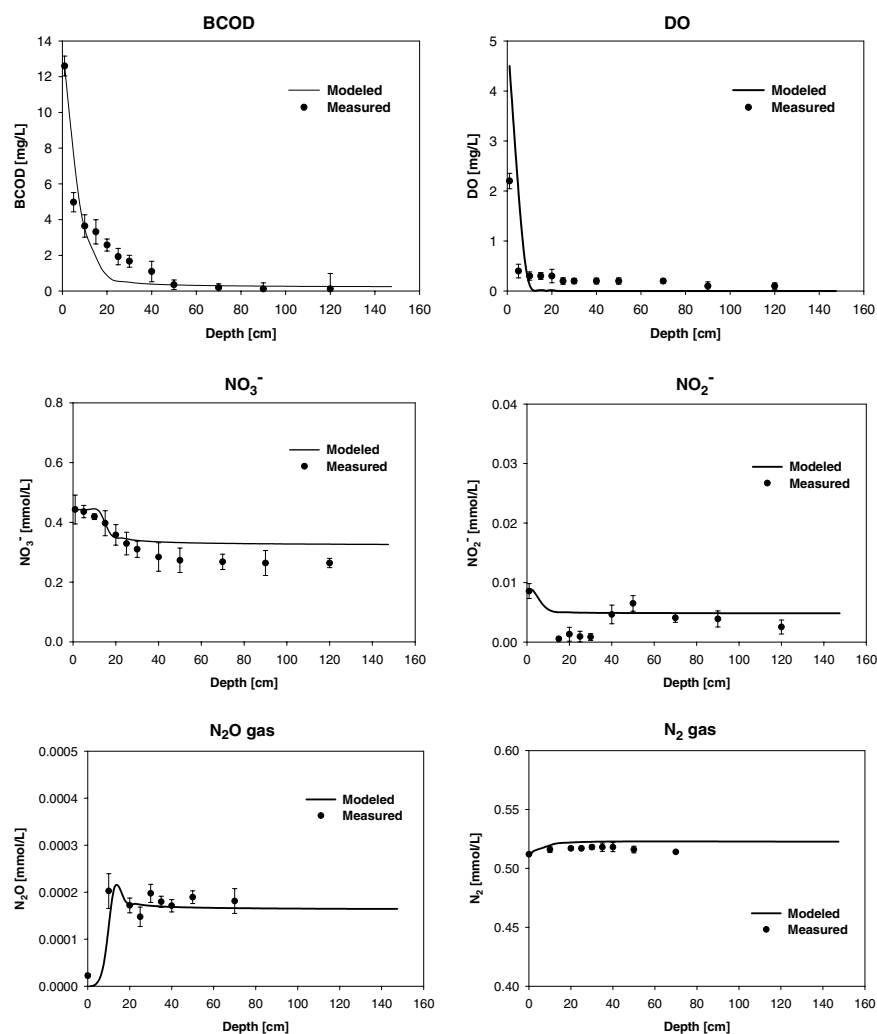


Figure 2. Comparison of model prediction with the measured data of the lysimeter experiment with the water application rate of 68.0 L/m² -day (data from Chung *et al.*, 2004)

soil surface and that the aerobic depth is limited only to a few decimeters from the soil surface. This can be compared with the study of Cho *et al.* (1997b) where the aerobic depth ranged 25–60 cm. This discrepancy was due to the low oxygen consumption caused by both low organic content and the low soil moisture content of the natural soil in the study of Cho *et al.* (1997b). The measured DO data had a trail behind the simulated concentration, which may suggest that the model overestimate the use of DO compared to the measurement and/or that the errors were involved in the measurement. Also it can be seen that the NO₃⁻ level decreases rapidly

Table IV. Parameter values changed through model calibration

Bioreaction parameters	Y	μ	k_d	K_S	a_{O_2}
	Yield coeff.	Maximum specific growth rate	Cell decay rate coeff.	Monod half coeff.	Oxygen consumption rate
	mg-MLVSS/ mg-BCOD	day ⁻¹	day ⁻¹	mg-BCOD L ⁻¹	mg-O ₂ consumed/ mg-COD removed
Aerobic	0.80	0.30	0.008	45.3	0.7
Anoxic	0.56	0.18	0.006	39.4	
Electron affinity coefficients	Q_{N_5} L/mmol of NO ₃ ⁻		Q_{N_3} L ² /mmol ² of NO ₂ ⁻		Q_{N_1} L/mmol of N ₂ O
		1		7000	1000

in the soil depth of 10–20 cm, which indicates active denitrification. The DO concentration in mmol/L decreased more rapidly than that of NO₃⁻ since the aerobic reaction is faster than the denitrifying one and the electron accepting capacity of oxygen is smaller than that of NO₃⁻-N. The NO₂⁻ and N₂O levels show that the NO₂⁻ level is 2 orders and the N₂O level is 3 orders lower than that of NO₃⁻ in the floodplain filtration. The high N₂O level in the soil depth of 10–20 cm is thought to be the result of the active denitrification in the zone (Cho *et al.*, 1997a,b).

A scenario analysis was performed to investigate the filtration phenomena of the contaminants when the influent BCOD concentration changed while that of NO₃⁻ was fixed. The BCOD concentrations simulated were 5, 10, 15, and 20 mg/L and that of the NO₃⁻-N was 3.5 mg/L (0.25 mmol/L) which is the typical NO₃⁻-N level in river waters in Korea. The relatively high BCOD levels assumed here was to investigate the applicability of this technique to the treatment of the secondary effluent of municipal wastewater. The influent DO level was set to 5.0 mg/L, and all other conditions were identical to those employed in Figure 2. The result is shown in Figure 3. From Figure 3 it can be seen that the BCOD concentration decreases rapidly in the vicinity of the soil surface regardless of the influent concentrations. With the influent BCOD level of 5 mg/L the BCOD goes deeper than with the other levels. This may be attributed to the less active development of biofilm with the lower BCOD level, resulting in the

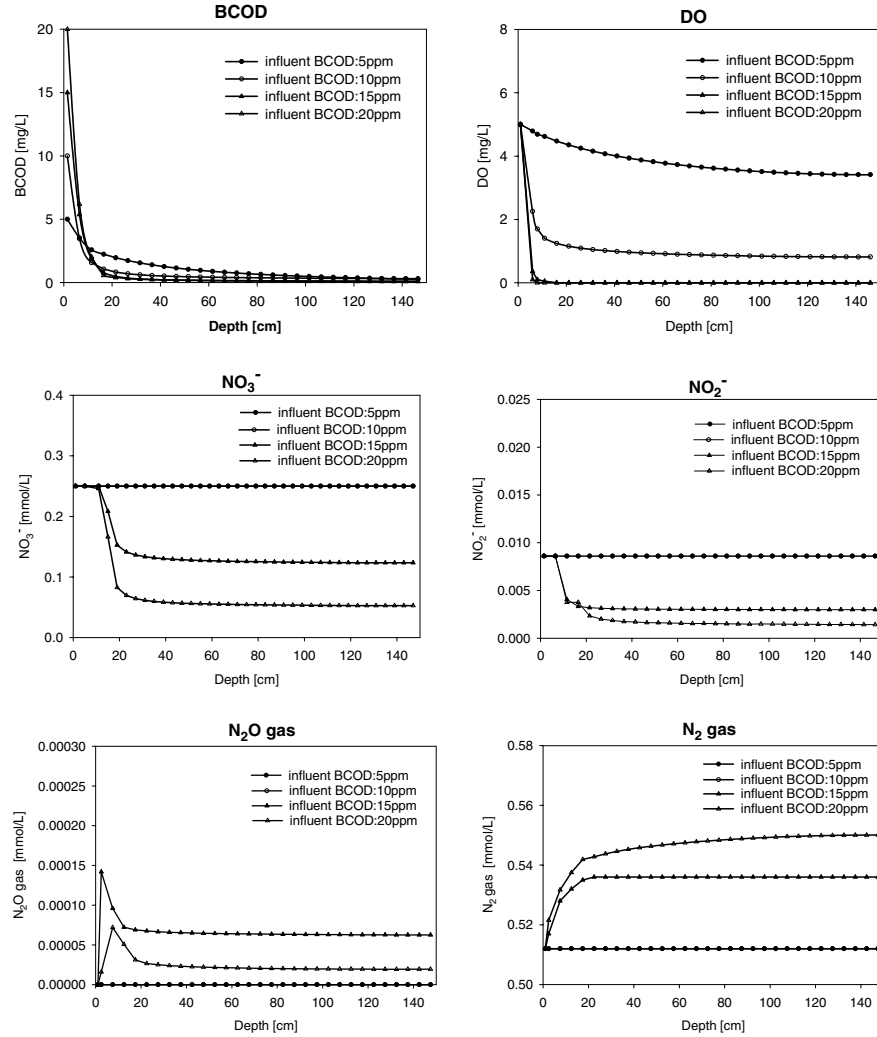


Figure 3. Effects of the influent BCOD concentration on floodplain filtration.

less BCOD consumption. It can also be seen that the DO and NO₃⁻ concentrations decrease as the BCOD level decreases along the soil depth. By comparing the BCOD, DO, and NO₃⁻ curves, however, it can be seen that there is some gap between the BCOD reduction and the electron acceptor consumption. This gap is due to the oxygen infiltration from the atmosphere through the soil gas phase. This infiltration rate increases along with the influent BCOD level since the oxygen concentration gradient is steeper with the higher influent BCOD level. With the influent BCOD levels of 11 mg/L or higher, despite the high oxygen infiltration rate, the DO level dropped drastically towards its exhaustion, and thereafter the NO₃⁻ level

decreased as expected. It can also be seen that in the active NO_3^- reduction zone the N_2O level is high, which is the result of the active denitrification (Cho *et al.*, 1997a,b). Most bioreactions occurred near the soil surface less than 40 cm in soil depth. The NO_3^- curves show that NO_3^- may leach if the BCOD level is not high enough. For the average river water in Korea the aerobic and denitrifying depths are expected not to exceed 20 and 40 cm from the soil surface, respectively. The N_2O emission rates were calculated as 0.000, 0.000, 0.036, and 0.004 $\text{mL/m}^2\text{-day}$ for the influent BCOD levels of 5, 10, 15, and 20 mg/L, respectively. Nitrogen mole balance showed that about 64% of the removed NO_3^- and NO_2^- escaped as N_2 emission and

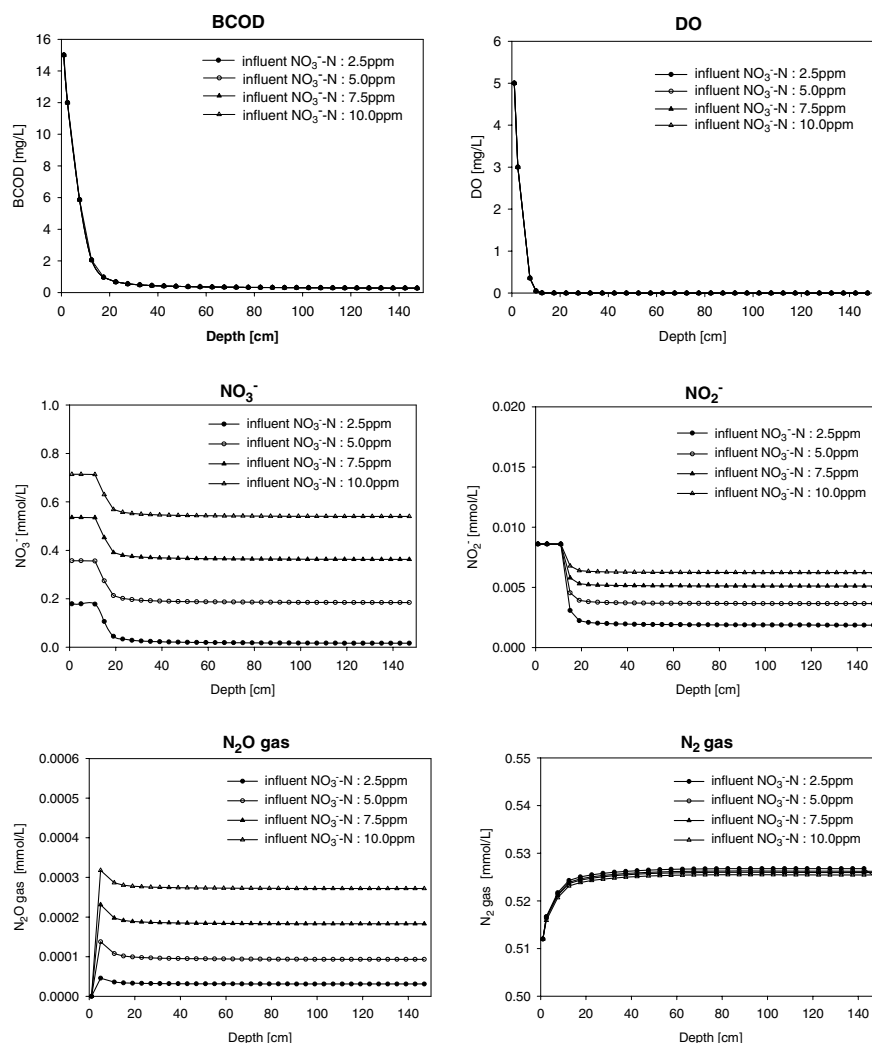


Figure 4. Effects of the influent NO_3^- concentration on floodplain filtration.

about 35% escaped as N_2 leaching. The N_2O leaching was 0.09% and its emission was 0.03%.

While the influent BCOD level was fixed as 15.0 mg/L, the model behavior was investigated in response to the change in NO_3^- concentration. The NO_3^- -N levels simulated were 2.5, 5.0, 7.5, and 10.0 mg/L. These relatively high levels were for the same reasons as in the case of BCOD change in Figure 3. All other conditions were identical to those employed in Figure 3. The results are shown in Figure 4. From Figure 4 it can be seen that the BCOD and DO profiles show similar patterns to those in Figures 2 and 3, and no actual difference was found according to the change in NO_3^- level. The NO_3^- curves show similar trend, however, there is a little difference in the NO_3^- consumption although it is not clearly recognizable. This difference can be explained by the differences in the increase of the NO_2^- and N_2O levels and in the N_2O emission rates. The NO_2^- and N_2O curves together with the NO_3^- curve indicate active denitrification in the soil depths of 10–20 cm. These are the unique characteristics that can be created only by the competitive Michaelis–Menten model. The nitrogen mole balance for Figure 4 is shown in Table V. As can be seen, N_2 emission was the primary path of nitrogen escape followed by N_2 leaching. The N_2O leaching and emission was 2–3 orders smaller than N_2 leaching. The N_2O emission rates were 0.012, 0.034, 0.058, and 0.079 mL/m²-day, respectively, for the influent NO_3^- -N levels of 2.5, 5.0, 7.5, and 10.0 mg/L, which showed that the N_2O emission increased along with the NO_3^- level. This relationship was also reported in a field study performed by Andrade *et al.* (2002).

Table V. Nitrogen mole balance in the floodplain filtration (application rate of 68 L/m²-day; influent BCOD of 15 mg/L; influent NO_3^- of varying concentrations)

NO_3^- -N (mg/L)	NO_3^-	NO_2^- (mmol/m ² -day)	N_2O (mmol/m ² -day)	N_2 (mmol/m ² -day)	§Total-N
2.5	+12.172	+0.585	–	+34.82	+82.39
	–1.156	–0.127	–0.0021(L)–0.0005(D)	–35.82(L)–2.79(D)	–78.51
5.0	+24.276	+0.585	–	+34.82	+94.49
	–12.576	–0.248	–0.0063(L)–0.0014(D)	–35.79(L)–2.67(D)	–89.76
7.5	+36.448	+0.585	–	+34.82	+106.66
	–24.707	–0.348	–0.0124(L)–0.0024(D)	–35.77(L)–2.61(D)	–101.84
10.0	+48.552	+0.585	–	+34.82	+118.77
	–36.760	–0.424	–0.0185(L)–0.0033(D)	–35.73(L)–2.31(D)	–113.31

§ about 5% of mole balance error occurred.

+: influx.

–: outflux.

L: Leaching.

D: Diffusion to the atmosphere.

In order to examine the possibility of the increase in the application rate, we studied the model behavior in response to varying filtration rates. The rates simulated were 68, 136, 204, and 272 L/m²-day, and their corresponding soil water contents were 0.340, 0.363, 0.377, and 0.385, respectively. The application rate of 68 L/m²-day was the maximum rate reported in the study of Chung *et al.* (2004). The influent BCOD level was set at 15 mg/L and all other parameters were kept identical to those in Figure 3. Figure 5 shows the result. As seen the BCOD level decreased rapidly with all application rates, and with the lower filtration rate the decrease rate was

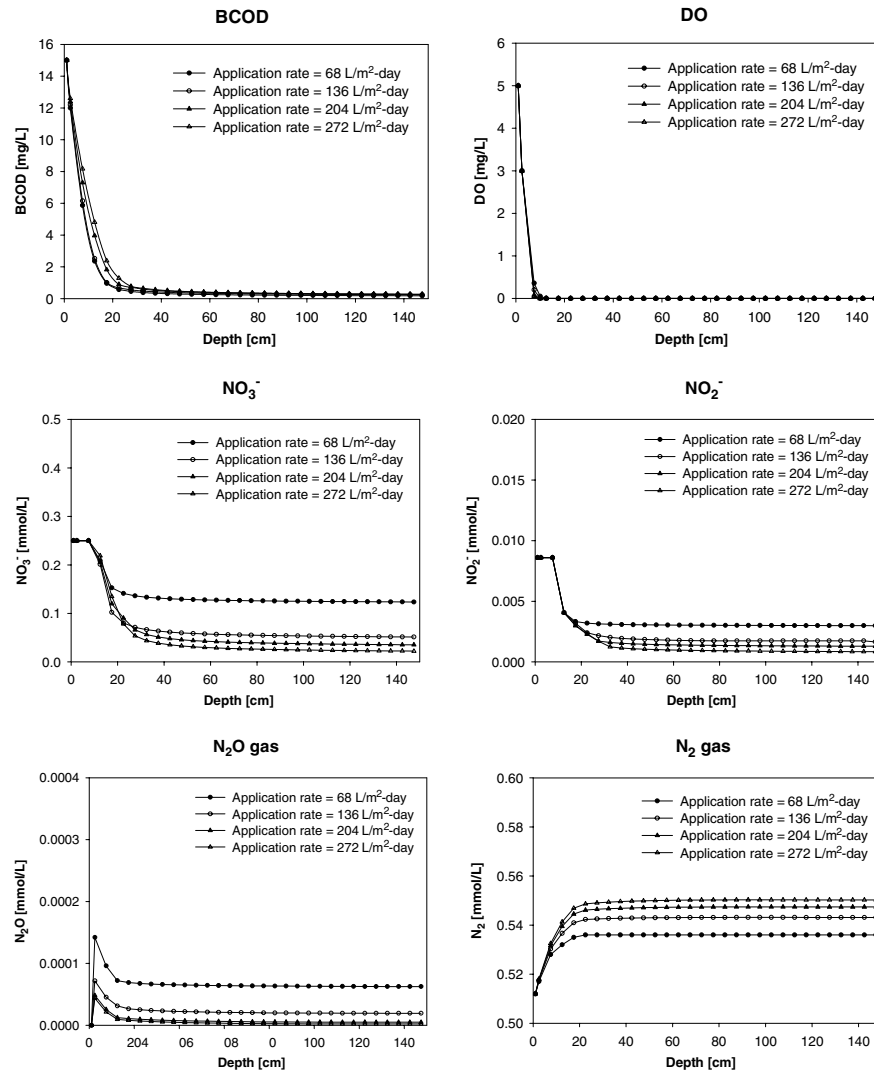


Figure 5. Effects of the application rate on floodplain filtration.

higher although the difference was not so large. This rate difference can be attributed to the increased bioreaction time in the soil enabled by the lower filtration rate. The DO curves show similar shape to those of other cases. The NO_3^- curves show that there is no practical difference according to the filtration rate up to the soil depth of 15 cm. In deep soil, however, these curves show differences and the concentration becomes lower as the filtration rate increases. This difference is due to the oxygen infiltration difference from the atmosphere through the soil gas phase. It is seen that as the application rate increases the performance of this technique also improves unless the NO_3^- is depleted. In other words, the N_2O emission and the NO_3^- and NO_2^- leaching decrease without significant sacrifice of the BCOD removal as the application rate increases. The N_2O emission rates were 0.036, 0.019, 0.012, and 0.010 mL/m²-day for the application rates of 68, 136, 204, and 272 L/m²-day, respectively.

Since the bioreactions can also take place in the saturated zone under the floodplain the minimum depth required for floodplain filtration can be regarded as the aerobic depth of the floodplain. Assuming the average organic content of the floodplain soil to be 1% in Korea (Kim *et al.*, 1998), we investigated the model behavior in response to the organic release rate (the result is not illustrated). If the release rate constant is larger than, for example, 1.0×10^{-8} kg-soil \times mg-BCOD/L-total soil volume-mg-organic matter-s, its effect on the BCOD and NO_3^- level in deep soil should not be neglected.

7. Conclusions

The mathematical model developed in this study described floodplain filtration favorably, and the filtration technique appears to be an effective way of removing both BCOD and NO_3^- simultaneously from the river water in Korea. It was found that a few decimeters of soil depth was enough to degrade most of the BCOD under varying conditions, and that the required aerobic and denitrifying depths were less than 20 and 40 cm from the soil surface, respectively. It was also found that there occurred oxygen infiltration through soil gas phase near the soil surface and this infiltration increased along with the influent BCOD level. Results also showed that the N_2O emission and the NO_3^- leaching increased as the influent NO_3^- level increased, and that as the application rate increased the performance of this technique improved without significant adverse effects such as the sacrifice of the BCOD removal and the increase of the N_2O emission. The N_2O emission in the floodplain filtration for most of the river waters in Korea was expected not to exceed 0.1 mL/m²-day. This technique was also expected to be applicable to the tertiary treatment of the municipal wastewater. The results of this study need to be tested again in a field scale

for further validation of the floodplain filtration technique. Although this study focused on the situation in Korea the results may be useful to similar watersheds around the world.

References

- Andrade, M.D., Prasher, S.O., Ahmad, D. and Clark, G.: 2002, Enhancement of denitrification by subsurface irrigation with sucrose enriched water, *Trans. ASAE*. **45**(6), 1749–1755.
- Baveye, P. and Valocchi, A.: 1989, An evaluation of mathematical models of the transport of biologically reacting solutes in saturated soils and aquifers, *Water Resour. Res.* **25**(6), 1313–1421.
- Bear, J.: 1972, *Dynamics of Fluids in Porous Media*, Dover Publications, New York.
- Burford, J. R. and Bremner, J. M.: 1975, Relationships between the denitrification capacities of soils and total, water-soluble and readily decomposable soil organic matter, *Soil Biol. Biochem.* **7**, 389–394.
- Characklis, W. G., Cunningham, F. B., Abedeen, F. and Crawford, D.: 1991, Influence of biofilm accumulation on porous media hydrodynamics, *Environ. Sci. Technol.* **25**(7), 1305–1311.
- Cho, C. M. and Mills, J. G.: 1979, Kinetic formulation of the denitrification process in soil, *Can. J. Soil Sci.* **59**, 249–257.
- Cho, C. M., Sakdinan, L. and Chang, C.: 1979, Denitrification intensity and capacity of three irrigated Alberta soils, *Soil Sci. Soc. Am. J.* **43**, 949–950.
- Cho, C. M.: 1982, Oxygen consumption and denitrification kinetics in soil, *Soil Sci. Soc. Am. J.* **46**, 756–762.
- Cho, C. M., Burton, D. L. and Chang, C.: 1997a, Kinetic formulation of oxygen consumption denitrification process in soil, *Can. J. Soil Sci.* **77**, 253–260.
- Cho, C. M., Burton, D. L. and Chang, C.: 1997b, Denitrification and fluxes of nitrogenous gases from soil under steady oxygen distribution, *Can. J. Soil Sci.* **77**, 261–269.
- Chung, J. B., Kim, S. H., Jeong, B. R. and Lee, Y. D.: 2004, Removal of nitrogen and organic matter from river water in a model system of floodplain filtration, *J. Environ. Qual.*, **33**, 1017–1023.
- Clement, T. P., Hooker, B. S. and Skeen, R. S.: 1996, Macroscopic models for predicting changes in saturated porous media properties caused by microbial growth, *Ground Water* **34**(5), 934–942.
- Collin, M. and Rasmuson, A.: 1988, A comparison of gas diffusivity models for unsaturated porous media, *Soil Sci. Soc. Am. J.* **52**, 1559–1565.
- Dean, J. A.: 1999, Lange's Handbook of Chemistry, 15th ed., McGraw-Hill, New York, USA.
- de Marsily, G.: 1986, *Quantitative Hydrogeology*, Academic Press, New York.
- Dendooven, L., Splatt, P. Anderson, J. M. and D. Scholefield.: 1994, Kinetic of the denitrification process in a soil under permanent pasture, *Soil Biol. Biochem.* **26**(3), 361–370.
- Domenico, P. A. and Schwartz, F. W.: 1990, *Physical and Chemical Hydrogeology*. Wiley, New York.
- Doussan, C., Poitevin, G. Ledoux, E. and Detay, M.: 1997, River bank filtration : modeling of the changes in water chemistry with emphasis on nitrogen species, *J. Contam. Hydrol.* **25**, 129–156.
- Guymon, G.L.: 1994, *Unsaturated Zone Hydrology*. Prentice-Hall, Englewood Cliffs, NJ.
- Hills, R.G., Porro, I., Hudson, D.B. and Wierenga P.J.: 1989, Modeling one-dimensional infiltration into very dry soils 1. Model development and evaluation, *Water Resour. Res.* **25**(6), 1259–1269.
- Kim, S. H. and Corapcioglu, M. Y.: 1996, A kinetic approach to modeling mobile bacteria facilitated groundwater contaminant transport, *Water Resour. Res.* **32**(2), 321–331.

- Kim, S. H., Kwon, O. O., Kong, I. C., Kim, I. J., Lee, C. H. and Park, Y. K.: 1998, A study on ground water recharge and nitrogen infiltration in a bank filtration site, *J. Korean Soc. Environ. Eng.* **20**(12), 1689–1703. [in Korean, English abstract.]
- Kim, S. H., Jeong, J. S. and Park, Y. K.: 2000, A study on the retardation of organic pollutant in bank filtration, *J. Korean Soc. Environ. Eng.* **22**(9), 1693–1705. [in Korean, English abstract.]
- Kim, S. H., Lee, Y. D. and Ro, H. M.: 2001, Partitioning of butachlor at air–water interface and transport in unsaturated sandy soils, *Environ. Eng. Sci.* **18**(2), 115–123.
- Kim, S. H., Chung, J. B., Ha, H. S. and Prasher, S. O.: 2003a, Tertiary treatment of municipal wastewater using unsaturated sandy soil, *Korean J. Environ. Agric.* 111–117. [in Korean, English abstract.]
- Kim, S. H., Chung, J. B., Jeong, B. R., Lee, Y. D. and Prasher, S. O.: 2003b, Electron affinity coefficients of nitrogen oxides and biodegradation kinetics in denitrification of contaminated stream water, *J. Environ. Qual.* **32**, 1474–1480.
- Kinzelbach, W., Schäfer, W. and Herzer, J.: 1991, Numerical modeling of natural and enhanced denitrification processes in aquifers, *Water Resour. Res.* **27**(6), 1123–1135.
- Lensing, H. J., Vogt, M. and Herrling, B.: 1994, Modeling of biologically mediated redox processes in the subsurface, *J. Hydrol.* **159**, 125–143.
- Millington and Shearer.: 1971, Diffusion in aggregated porous media, *Soil Sci.* **111**, 372–378.
- Molz, F. J., Widdowson, M. A. and Benefield, L. D.: 1986, Simulation of Microbial growth dynamics coupled to nutrient and oxygen transport in porous media, *Water Resour. Res.* **22**(8), 1207–1216.
- Patrick, W. H., Jr. and Jugsujinda, A.: 1992, Sequential reduction oxidation of inorganic nitrogen, manganese and iron in flooded soil, *Soil Sci. Soc. Am. J.* **56**, 1071–1073.
- Qiyang, Y. and Boersma, L.: 1992, Dynamic oxygen and carbon dioxide exchange between soil and atmosphere, I. Model development. *Soil Sci. Soc. Am. J.* **56**, 1695–1702.
- Ramalho, R. S.: 1983, *Introduction to Wastewater Treatment Processes*, Academic press, New York.
- Rittmann, B. E.: 1993, The significance of biofilms in porous media, *Water Resour. Res.* **29**, 2195–2202.
- Stanford, G., Dzienia, S. and Vanderpol, R. A.: 1975, Effect of temperature on denitrification rate in soils, *Soil Sci. Soc. Am. Proc.* **39**, 867–870.
- Sung, C. D., Ahn, K. H., Lee, Y. H., Kong, I. C. and Kim, S. H.: 2003, A Study on the water quality improvement through riverbed filtration – Model development and parameter determination, *J. Korean Soc. Environ. Eng.* **25**(4), 486–494. [in Korean, English abstract.]
- van Genuchten, M. Th. and Alves, W. J.: 1982, Analytical solutions of the one-dimensional convective–dispersive solute transport equation, Agricultural Research Service, U.S. Department of Agriculture, *Technical Bulletin* No. 1661.