



Effect of dissolved organic matter and bacteria on contaminant transport in riverbank filtration

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Abstract

A mathematical model for the transport of hydrophobic organic contaminants in an aquifer under simplistic riverbank filtration conditions is developed. The model considers a situation where contaminants are present together with dissolved organic matter (DOM) and bacteria. The aquifer is conceptualized as a four-phase system: two mobile colloidal phases, an aqueous phase, and a stationary solid phase. An equilibrium approach is used to describe the interactions of contaminants with DOM, bacteria, and solid matrix. The model is composed of bacterial transport equation and contaminant transport equation. Numerical simulations are performed to examine the contaminant transport behavior in the presence of DOM and bacteria. The simulation results illustrate that contaminant transport is enhanced markedly in the presence of DOM and bacteria, and the impact of DOM on contaminant mobility is greater than that of bacteria under examined conditions. Sensitivity analysis demonstrates that the model is sensitive to changes of three lumped parameters: K_1^+ (total affinity of stationary solid phase to contaminants), K_2^+ (total affinity of DOM to contaminants), and K_3^+ (total affinity of bacteria to contaminants). In a situation where contaminants exist simultaneously with DOM and bacteria, contaminant transport is mainly affected by a ratio of $K_1^+/K_2^+/K_3^+$, which can vary with changes of equilibrium distribution coefficient of contaminants and/or colloidal concentrations. In riverbank filtration, the influence of DOM and bacteria on the transport behavior of contaminants should be accounted to accurately predict the contaminant mobility.

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1. Introduction

Riverbank filtration is a natural process, using alluvial aquifers to remove contaminants and pathogens in river water for the production of drinking water (Fig. 1). It is widely applied in the European countries such as Germany, Netherlands, France, Switzerland, and Hungary. In Germany, riverbank filtration has been used along the Rhine River for many years. Kuehn and Mueller (2000) reported that riverbank filtration/infiltration constitutes nearly 16% of drinking water production in Germany. In the Netherlands, riverbank filtration is also practiced along the Rhine River (van der Kooij et al., 1985). In France,

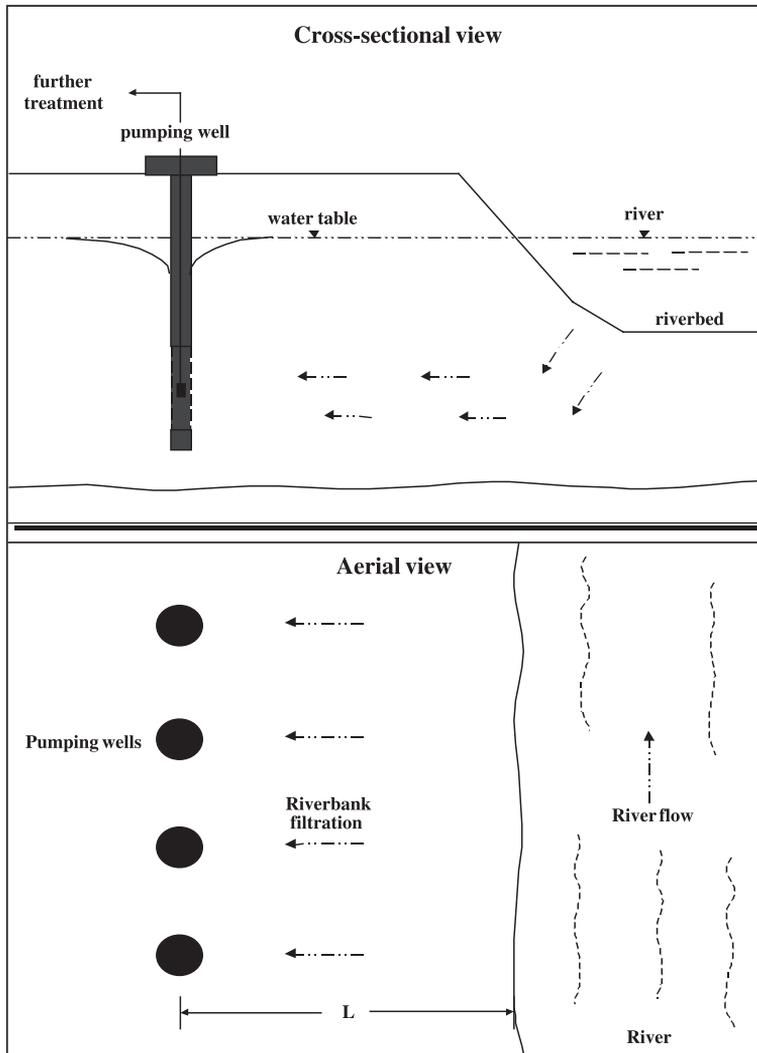


Fig. 1. Schematic diagram of riverbank filtration.

alluvial aquifers are well developed along the Deûle River, the Lot River, the Rhône River, and the Seine River, and the riverbank filtration is performed along those rivers for the drinking water production (Bourg and Bertin, 1994; Doussan et al., 1997). In the lower Glatt Valley, Switzerland, the Glatt River recharges the shallow aquifer, mainly composed of gravel and sands, by infiltrating into the top layer of saturated aquifer (von Gunten et al., 1991). Recently, riverbank filtration is applied in the United States as a treatment technology due to its removal efficiency and cost-effectiveness in drinking water treatment (Ray et al., 2002).

In riverbank filtration, the physical, chemical, and microbiological qualities of bank-filtered water primarily depend on the quality of river water. In a situation where chemical pollution is not serious in river, bank-filtered water can be used directly as drinking water after disinfection. However, if contamination of river is serious due to chemicals discharged from industries, additional treatments are required to achieve drinking water standards. The quality of bank-filtered water is also affected by the riverbed sediment, the aquifer media, the infiltration velocity, and the residence time in the aquifer (Literathy and László, 1996).

The organic compounds discharged from chemical plants and industries are the major contaminants to cause river pollution and subsequently to impact on the quality of the bank-filtered water (Sontheimer, 1980; Piet and Zoeteman, 1980). The behavior of organic contaminants in riverbank filtration has been studied by several researchers (Schwarzenbach and Westall, 1981; Schwarzenbach et al., 1983; Schellenberg et al., 1984; Kuhn et al., 1985; Ahel et al., 1996). In riverbank filtration, the fate and transport of organic contaminants are mainly affected by microbial degradation, sorption to solid matrix, and attachment to colloidal particles.

Colloids are widely present in surface (e.g., rivers) and subsurface (e.g., groundwater) aquatic environments. They can be classified into three major types (Mills et al., 1991): (i) biocolloids, (ii) inorganic colloids, (iii) organic colloids. Colloids can play the role of mobile carriers in facilitating the contaminant transport (Corapcioglu and Jiang, 1993). Colloids can enhance the mobility of inorganic contaminants such as metals and radionuclides (Grolimund et al., 1996; Noell et al., 1998), and organic contaminants such as polycyclic aromatic hydrocarbons (PAHs) and dichlorodiphenyltrichloroethane (DDT) in porous media (Jenkins and Lion, 1993; Saiers and Hornberger, 1996). In riverbank filtration, mobility of contaminants may be enhanced by mobile colloids such as dissolved organic matter (DOM) and mobile bacteria.

DOM is a complex molecule, composed of humic (humic and fulvic acids) and nonhumic substances (amino acids and carbohydrates) and abundant in aquatic environments (Frimmel, 1998). In riverbank filtration, most of DOM removal occurs around the river-aquifer interface by physical and biochemical processes. Biodegradable fraction of DOM would be degraded by bacteria while refractory fraction removed by adsorption on solid phase (Marmonier et al., 1995). DOM may enhance the mobility of contaminants through aquifers. Enfield et al. (1989) have demonstrated that hydrophobic organic compounds could pass through soils faster when they partitioned into DOM. Magee et al. (1991) have shown that hydrophobic compounds had strong affinity to DOM and moved faster in the presence of DOM. Liu and Amy (1993) have reported that DOM suspended in the aqueous phase could facilitate the transport of PAHs.

The fate and transport of contaminants may also be affected by the presence of mobile bacteria. The concentration of organic contaminants may be reduced by bacterial utilization, and the transport behavior of contaminants may be altered due to their sorption onto bacteria. The attachment of contaminants onto bacteria, called biosorption, is a well-known phenomenon (Tsezos and Seto, 1986; Bellin and Rao, 1993). As mobile colloids, bacteria can carry hydrophobic organic contaminants sorbed onto them. Especially when bacteria have high mobility, contaminants may be transported further than expected, resulting in serious groundwater contamination. Lindqvist and Enfield (1992) have reported that DDT and hexachlorobenzene could adsorb onto bacteria, and hence their transport could be facilitated in groundwater. Jenkins and Lion (1993) also have confirmed that the mobility of hydrophobic organic compounds such as PAHs could be enhanced due to their sorption to mobile bacteria in porous media.

The objective of this study was to investigate the transport behavior of hydrophobic organic contaminants in an aquifer under simplistic riverbank filtration conditions where contaminants were present simultaneously with DOM and bacteria and uniform influent water quality was imposed. A mathematical model is developed based on the conceptualization of the aquifer as a four-phase system: two mobile colloidal phases, an aqueous phase, and a stationary solid phase (Fig. 2). An equilibrium approach is used to describe the interactions of contaminants with DOM, bacteria, and solid matrix. A fully implicit

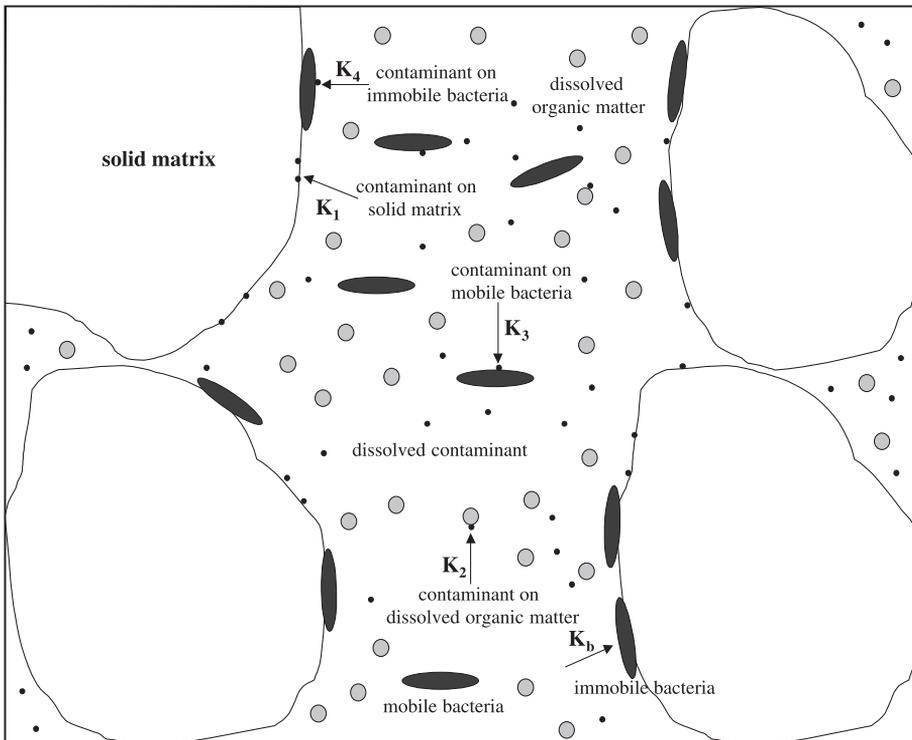


Fig. 2. Conceptual diagram for contaminant transport in the presence of DOM and bacteria.

finite difference method is applied to obtain a numerical solution of the dimensionless form of proposed model. Numerical simulations are performed to examine the contaminant transport in the presence of DOM and bacteria. Sensitivity analysis is conducted to observe the effect of key model parameters on the contaminant transport behavior.

2. Mathematical model development

2.1. DOM and bacterial transport

McCarthy et al. (1996) have reported that DOM transport could be considered as that of a conservative solute in the aquifer where an equilibrium condition between DOM and organic matter on the solid phase was not disrupted or where the aquifer binding sites for organic matter were saturated due to continuous input. It is assumed in this study that the aquifer is saturated with organic matter due to continuous input of DOM from river water.

The mass balance equation for bacteria suspended in the aqueous phase of saturated porous media may be described as:

$$\frac{\partial(\theta C_b)}{\partial t} = -\nabla \cdot [-D_b \nabla(\theta C_b) + v_w \theta C_b] - Q_{bs} + Q_{gm} - Q_{dm} + Q_{go} \quad (1)$$

where θ is the water content ($\theta = n - \sigma_b$), n is the porosity, σ_b is the volumetric fraction of bacteria attached onto solid matrix (volume of bacteria deposited per unit total volume of porous media), C_b is the concentration of bacteria suspended in the aqueous phase, D_b is the hydrodynamic dispersion coefficient of bacteria ($L^2 T^{-1}$), and v_w is the pore-water velocity ($L T^{-1}$). Q_{bs} represents the mass transfer of bacteria between the aqueous and solid phases (mass of bacteria per unit volume of porous media per unit time). Q_{gm} denotes the growth rate of bacteria suspended in the aqueous phase with contaminant as a food source. Q_{dm} denotes the decay rate of bacteria suspended in the aqueous phase. Q_{go} indicates the growth rate of bacteria suspended in the aqueous phase with DOM as a food source. These terms will be described further below.

The mass balance equation for bacteria captured on the surfaces of solid matrix may be described as:

$$\frac{\partial(\rho_b \sigma_b)}{\partial t} = Q_{bs} + Q_{gi} - Q_{di} \quad (2)$$

where ρ_b is the density of bacteria. Q_{gi} and Q_{di} denote the growth and decay rates of bacteria captured on solid matrix, respectively.

The attachment of bacteria on solid matrix may be described as an equilibrium-controlled process (Matthess et al., 1988). If the deposition of bacteria on the surfaces of solid matrix is a linear and reversible process, the volumetric fraction of bacteria captured on solid matrix may be given as:

$$\sigma_b = K_b C_b \quad (3)$$

where K_b is the linear equilibrium distribution coefficient for bacteria between the aqueous phase and solid matrix ($L^3 M^{-1}$).

2.2. Contaminant transport

In the presence of DOM and mobile bacteria in groundwater, contaminants may be dissolved in the aqueous phase and sorbed on solid matrix, DOM or bacteria. The mass balance equation for contaminants dissolved in the aqueous phase may be expressed as:

$$\frac{\partial(\theta C_c)}{\partial t} = -\nabla \cdot [-D_c \nabla(\theta C_c) + v_w \theta C_c] - Q_{cs} - Q_{cd} - Q_{cbm} - Q_{cbi} - Q_a \quad (4)$$

where C_c is the concentration of dissolved contaminants, D_c is the hydrodynamic dispersion coefficient of contaminants ($L^2 T^{-1}$), Q_{cs} is the mass transfer of contaminants between the aqueous phase and solid matrix (mass of contaminants per unit volume of porous media per unit time), Q_{cd} denotes the mass transfer of contaminants between the aqueous phase and DOM, Q_{cbm} indicates the mass transfer of contaminants between the aqueous phase and mobile bacteria, Q_{cbi} represents the mass transfer of contaminants between the aqueous phase and immobile bacteria, and Q_a is the utilization rate of contaminants dissolved in the aqueous phase.

The mass balance equation for contaminants sorbed on solid matrix may be presented as:

$$\frac{\partial(\rho_s \sigma_{cs})}{\partial t} = Q_{cs} - Q_s \quad (5)$$

ρ_s is the dry bulk density of solid matrix, σ_{cs} is the mass fraction of contaminants sorbed on solid matrix (mass of contaminants sorbed per unit solid mass of porous media), and Q_s is the utilization rate of contaminants sorbed on solid matrix. Sorption of hydrophobic organic compounds on solid matrix is controlled by hydrophobic partitioning (Brusseau and Rao, 1989). If sorption of contaminants on solid matrix can be represented by a linear equilibrium isotherm, the mass fraction of contaminants sorbed on solid matrix may be given as:

$$\sigma_{cs} = K_1 C_c \quad (6)$$

where K_1 is the linear equilibrium distribution coefficient of contaminants between the aqueous phase and solid matrix ($L^3 M^{-1}$).

The mass balance equation for contaminants sorbed to DOM may be presented as:

$$\frac{\partial(\theta C_d \sigma_{cd})}{\partial t} = -\nabla \cdot [-D_d \nabla(\theta C_d \sigma_{cd}) + v_w \theta C_d \sigma_{cd}] + Q_{cd} - Q_d \quad (7)$$

where C_d is the concentration of DOM in the aqueous phase, σ_{cd} is the mass fraction of contaminants attached to DOM (mass of contaminants sorbed on DOM per unit mass of DOM), D_d is the hydrodynamic dispersion coefficient of DOM ($L^2 T^{-1}$), and Q_d is the utilization rate of contaminants sorbed on DOM. As noted earlier, in riverbank filtration,

the aquifer is generally saturated with DOM. Therefore, for practical purposes of riverbank filtration, we can assume constant DOM concentration in groundwater. Assuming that attachment of contaminants to DOM is an equilibrium-controlled process, the mass fraction of contaminants sorbed on DOM can be described with a linear equilibrium isotherm as:

$$\sigma_{cd} = K_2 C_c \quad (8)$$

where K_2 is the linear equilibrium distribution coefficient of contaminants between the aqueous phase and DOM ($L^3 M^{-1}$).

The mass balance equation for contaminants attached to mobile bacteria may be expressed as:

$$\frac{\partial(\theta C_b \sigma_{cbm})}{\partial t} = -\nabla \cdot [-D_b \nabla(\theta C_b \sigma_{cbm}) + v_w \theta C_b \sigma_{cbm}] + Q_{cbm} - Q_{bm} \quad (9)$$

where σ_{cbm} is the mass fraction of contaminants attached to mobile bacteria (mass of contaminants sorbed on mobile bacteria per unit mass of mobile bacteria) and Q_{bm} is the utilization rate of contaminants sorbed on mobile bacteria. The mass balance equation for contaminants attached to immobile bacteria may be written as:

$$\frac{\partial(\rho_b \sigma_b \sigma_{cbi})}{\partial t} = Q_{cbi} - Q_{bi} \quad (10)$$

where σ_{cbi} is the mass fraction of contaminants attached to immobile bacteria (mass of contaminants sorbed on immobile bacteria per unit mass of immobile bacteria) and Q_{bi} is the utilization rate of contaminants sorbed on immobile bacteria.

With an assumption that sorption of contaminants on bacteria is an equilibrium-controlled process, the mass fractions of contaminants attached to mobile and immobile bacteria can be presented using a linear equilibrium isotherm as:

$$\sigma_{cbm} = K_3 C_c \quad (11)$$

$$\sigma_{cbi} = K_4 C_c \quad (12)$$

where K_3 is the linear equilibrium distribution coefficient of contaminants between the aqueous phase and mobile bacteria ($L^3 M^{-1}$) and K_4 is the linear equilibrium distribution coefficient of contaminants between the aqueous phase and immobile bacteria ($L^3 M^{-1}$).

2.3. Bacterial growth and decay and contaminant utilization

If bacteria can utilize contaminants as a growth substrate in porous media, the growth rate of mobile bacteria may be expressed by the following kinetic expression as:

$$Q_{gm} = \mu_g \theta C_b \quad (13)$$

where μ_g is the specific growth rate of mobile bacteria (T^{-1}). The specific growth rate of bacteria can be described with the Monod equation. Assuming that contaminant concen-

tration in the aqueous phase is low ($K_s \gg C_c$), the growth rate of mobile bacteria with contaminants as a food source may be presented as (Kim and Corapcioglu, 1996):

$$Q_{gm} = \mu \left(C_c + \frac{\rho_s \sigma_{cs}}{\theta} \right) \theta C_b \quad (14)$$

Note that $\mu \approx \mu_{\max}/K_s$, where μ_{\max} is the maximum growth rate (T^{-1}) and K_s is the half-saturation constant (ML^{-3}). Furthermore, if bacteria can utilize contaminants sorbed on bacteria, DOM, and solid matrix, Eq. (14) may be rewritten as:

$$Q_{gm} = \mu \left(C_c + \frac{\rho_s \sigma_{cs}}{\theta} + C_d \sigma_{cd} + C_b \sigma_{cbm} \right) \theta C_b \quad (15)$$

After following the same sequence of derivation and assuming that the specific growth rate of immobile bacteria is equal to that of mobile bacteria, the growth rate of immobile bacteria may be expressed as:

$$Q_{gi} = \mu \left(C_c + \frac{\rho_s \sigma_{cs}}{\theta} + \frac{\rho_b \sigma_b \sigma_{cbi}}{\theta} \right) \rho_b \sigma_b \quad (16)$$

The growth rate of bacteria with DOM as a food source may be expressed as first-order kinetic expression as (Borden and Bedient, 1986):

$$Q_{go} = k_o Y \theta C_d \quad (17)$$

where k_o is the first-order decay rate coefficient for DOM (T^{-1}) and Y is the yield coefficient (mass of bacteria formed per unit mass of food source utilized).

The decay rates of mobile and immobile bacteria may be described by the first-order kinetic expressions as:

$$Q_{dm} = k_{dm} \theta C_b \quad (18)$$

$$Q_{di} = k_{di} \rho_b \sigma_b \quad (19)$$

where k_{dm} and k_{di} are the decay rate coefficients for mobile and immobile bacteria (T^{-1}), respectively. The utilization rates of contaminants present in the aqueous phase or on solid matrix may be written as:

$$Q_a = \frac{\mu C_c}{Y} (\theta C_b + \rho_b \sigma_b) \quad (20)$$

$$Q_s = \frac{\mu \rho_s \sigma_{cs}}{Y \theta} (\theta C_b + \rho_b \sigma_b) \quad (21)$$

$$Q_d = \frac{\mu C_d \sigma_{cd}}{Y} (\theta C_b) \quad (22)$$

$$Q_{bm} = \frac{\mu C_b \sigma_{cbm}}{Y} (\theta C_b) \quad (23)$$

$$Q_{bi} = \frac{\mu \rho_b \sigma_b \sigma_{cbi}}{Y} (\rho_b \sigma_b) \quad (24)$$

Eqs. (20) and (21) represent the utilization rates of contaminants dissolved in the aqueous phase and sorbed on solid matrix, respectively. Eqs. (22)–(24) denote the utilization rates of contaminants sorbed on DOM, mobile bacteria, and immobile bacteria, respectively.

2.4. One-dimensional bacterial transport equation

Substitution of Eq. (2) into Eq. (1) yields the bacterial transport equation as:

$$\frac{\partial(\theta C_b)}{\partial t} + \frac{\partial(\rho_b \sigma_b)}{\partial t} = -\nabla \cdot [-D_b \nabla(\theta C_b) + v_w \theta C_b] + Q_{gm} + Q_{gi} - Q_{dm} - Q_{di} + Q_{go} \quad (25)$$

Then, substitution of Eqs. (15)–(19) into Eq. (25) and rearrangement generates:

$$\begin{aligned} \frac{\partial}{\partial t}(\theta C_b + \rho_b \sigma_b) &= -\nabla \cdot [-D_b \nabla(\theta C_b) + v_w \theta C_b] \\ &+ \mu \left(C_c + \frac{\rho_s \sigma_{cs}}{\theta} + C_d \sigma_{cd} + C_b \sigma_{cbm} \right) \theta C_b \\ &+ \mu \left(C_c + \frac{\rho_s \sigma_{cs}}{\theta} + \frac{\rho_b \sigma_b \sigma_{cbi}}{\theta} \right) \rho_b \sigma_b - k_{dm} \theta C_b - k_{di} \rho_b \sigma_b \\ &+ k_o Y \theta C_d \end{aligned} \quad (26)$$

After substitution of Eqs. (3), (6), (8), (11), and (12) into Eq. (26), the one-dimensional transport equation for bacteria can be expressed as:

$$\begin{aligned} \frac{\partial(R_b \theta C_b)}{\partial t} &= -\frac{\partial}{\partial x} \left[-D_b \frac{\partial(\theta C_b)}{\partial x} + v_w \theta C_b \right] \\ &+ \mu \left(C_c + \frac{\rho_s K_1 C_c}{\theta} + C_d K_2 C_c + C_b K_3 C_c \right) \theta C_b \\ &+ \mu \left(C_c + \frac{\rho_s K_1 C_c}{\theta} + \frac{\rho_b K_b C_b K_4 C_c}{\theta} \right) \rho_b K_b C_b - k_{dm} \theta C_b \\ &- k_{di} \rho_b K_b C_b + k_o Y \theta C_d \end{aligned} \quad (27)$$

where R_b is a retardation factor for bacteria

$$R_b = \left(1 + \frac{\rho_b K_b}{\theta} \right) \quad (28)$$

2.5. One-dimensional contaminant transport equation

Substitution of Eqs. (5), (7), (9), and (10) into Eq. (4) yields the equation for contaminant transport as:

$$\begin{aligned} & \frac{\partial(\theta C_c)}{\partial t} + \frac{\partial(\rho_s \sigma_{cs})}{\partial t} + \frac{\partial(\theta C_d \sigma_{cd})}{\partial t} + \frac{\partial(\theta C_b \sigma_{cbm})}{\partial t} + \frac{\partial(\rho_b \sigma_b \sigma_{cbi})}{\partial t} \\ & = -\nabla \cdot [-D_c \nabla(\theta C_c) + v_w \theta C_c] - \nabla \cdot [-D_d \nabla(\theta C_d \sigma_{cd}) + v_w \theta C_d \sigma_{cd}] \\ & \quad - \nabla \cdot [-D_b \nabla(\theta C_b \sigma_{cbm}) + v_w \theta C_b \sigma_{cbm}] - Q_a - Q_s - Q_d - Q_{bm} - Q_{bi} \end{aligned} \quad (29)$$

Substitution of Eqs. (20)–(24) into Eq. (29) and rearrangement produces:

$$\begin{aligned} & \frac{\partial}{\partial t} (\theta C_c + \rho_s \sigma_{cs} + \theta C_d \sigma_{cd} + \theta C_b \sigma_{cbm} + \rho_b \sigma_b \sigma_{cbi}) \\ & = -\nabla \cdot [-D_c \nabla(\theta C_c) + v_w \theta C_c] - \nabla \cdot [-D_d \nabla(\theta C_d \sigma_{cd}) + v_w \theta C_d \sigma_{cd}] \\ & \quad - \nabla \cdot [-D_b \nabla(\theta C_b \sigma_{cbm}) + v_w \theta C_b \sigma_{cbm}] - \frac{\mu C_c}{Y} (\theta C_b + \rho_b \sigma_b) \\ & \quad - \frac{\mu \rho_s \sigma_{cs}}{Y \theta} (\theta C_b + \rho_b \sigma_b) - \frac{\mu C_d \sigma_{cd}}{Y} (\theta C_b) - \frac{\mu C_b \sigma_{cbm}}{Y} (\theta C_b) - \frac{\mu \rho_b \sigma_b \sigma_{cbi}}{Y \theta} (\rho_b \sigma_b) \end{aligned} \quad (30)$$

Then, substitution of Eqs. (3), (6), (8), (11), and (12) into Eq. (30) yields:

$$\begin{aligned} & \frac{\partial}{\partial t} [(\theta + \rho_s K_1 + \theta K_2 C_d + \theta K_3 C_b + \rho_b K_b K_4 C_b) C_c] \\ & = -\nabla \cdot [-D_c \nabla(\theta C_c) + v_w \theta C_c] - \nabla \cdot [-D_d \nabla(\theta K_2 C_d C_c) + v_w \theta K_2 C_d C_c] \\ & \quad - \nabla \cdot [-D_b \nabla(\theta K_3 C_b C_c) + v_w \theta K_3 C_b C_c] - \frac{\mu C_c}{Y} (\theta C_b + \rho_b K_b C_b) \\ & \quad - \frac{\mu \rho_s K_1 C_c}{Y \theta} (\theta C_b + \rho_b K_b C_b) - \frac{\mu C_d K_2 C_c}{Y} (\theta C_b) - \frac{\mu C_b K_3 C_c}{Y} (\theta C_b) \\ & \quad - \frac{\mu \rho_b K_b C_b K_4 C_c}{Y \theta} (\rho_b K_b C_b) \end{aligned} \quad (31)$$

After following the derivation for R_c (contaminant retardation factor) described in Johnson et al. (1998), the one-dimensional transport equation for contaminants can be finalized as:

$$\begin{aligned} \frac{\partial(R_c \theta C_c)}{\partial t} & = -\frac{\partial}{\partial x} \left[-D_c \frac{\partial(\theta C_c)}{\partial x} + v_w \theta C_c \right] - \frac{\partial}{\partial x} \left[-D_d \frac{\partial(\theta K_2 C_d C_c)}{\partial x} + v_w \theta K_2 C_d C_c \right] \\ & \quad - \frac{\partial}{\partial x} \left[-D_b \frac{\partial(\theta K_3 C_b C_c)}{\partial x} + v_w \theta K_3 C_b C_c \right] - \frac{\mu C_c}{Y} (\theta C_b + \rho_b K_b C_b) \\ & \quad - \frac{\mu \rho_s K_1 C_c}{Y \theta} (\theta C_b + \rho_b K_b C_b) - \frac{\mu C_d K_2 C_c}{Y} (\theta C_b) - \frac{\mu C_b K_3 C_c}{Y} (\theta C_b) \\ & \quad - \frac{\mu \rho_b K_b C_b K_4 C_c}{Y \theta} (\rho_b K_b C_b) \end{aligned} \quad (32)$$

where R_c is represented as

$$R_c = \left[1 + \frac{(\rho_s K_1 + K_4 K_b \rho_b C_b)}{\theta(1 + K_2 C_d + K_3 C_b)} \right] \tag{33}$$

2.6. Dimensional analysis

The result of contaminant transport model can be generalized by grouping individual parameters of the phenomenon in dimensionless numbers. These numbers reveal the functional dependence of each set of dimensionless groups in response to orders of magnitude changes in one dimensionless group to the other. Therefore, in this study we present a dimensional analysis for the governing equations by employing the following dimensionless variables:

$$\begin{aligned} X = \frac{x}{L}; T = \frac{tv_w}{L}; \theta^+ = \frac{\theta}{n}; C_b^+ = \frac{C_b}{C_{b0}}; C_c^+ = \frac{C_c}{C_{c0}}; K_1^+ = \frac{\rho_s K_1}{n}; K_2^+ = K_2 C_d; \\ K_3^+ = K_3 C_{b0}; K_4^+ = K_4 C_{b0}; K_b^+ = \frac{\rho_b K_b}{n} \end{aligned} \tag{34}$$

where C_{b0} is the influent concentration of bacteria at $x=0$, C_{c0} is the influent concentration for contaminants at $x=0$, and L is the distance between the river and the pumping station (see Fig. 1). In addition, the following dimensional parameters (Peclet number and Damköhler numbers) are used:

$$Pe = \frac{v_w L}{D_c}; Da_1 = \frac{Lk_{dm}}{v_w}; Da_2 = \frac{Lk_{di}}{v_w}; Da_3 = \frac{L\mu C_{c0}}{v_w}; Da_4 = \frac{L\mu C_{b0}}{v_w Y}; Da_5 = \frac{Lk_0}{v_w} \tag{35}$$

From the dimensional analysis, (Eqs. (27), (28), (32), and (33) can be rewritten in terms of dimensionless parameters and variables as:

$$\begin{aligned} \frac{\partial(R_b^+ C_b^+)}{\partial T} = \frac{1}{Pe} \frac{\partial^2 C_b^+}{\partial X^2} - \frac{\partial C_b^+}{\partial X} + Da_3 C_c^+ \left(1 + \frac{K_1^+}{\theta^+} + K_2^+ + K_3^+ C_b^+ \right) C_b^+ + Da_3 C_c^+ \\ \times \left(1 + \frac{K_1^+}{\theta^+} + \frac{K_b^+ C_b^+ K_4^+}{\theta^+} \right) K_b^+ C_b^+ - Da_1 C_b^+ - Da_2 K_b^+ C_b^+ + Da_5 C_b^+ \end{aligned} \tag{36}$$

$$R_b^+ = \left(1 + \frac{K_b^+}{\theta^+} \right) \tag{37}$$

$$\begin{aligned} \frac{\partial(R_c^+ C_c^+)}{\partial T} = & \frac{1}{Pe} \frac{\partial^2 C_c^+}{\partial X^2} - \frac{\partial C_c^+}{\partial X} + \frac{1}{Pe} \frac{\partial^2 (K_2^+ C_c^+)}{\partial X^2} - \frac{\partial (K_2^+ C_c^+)}{\partial X} + \frac{1}{Pe} \frac{\partial^2 (C_b^+ K_3^+ C_c^+)}{\partial X^2} \\ & - \frac{\partial (C_b^+ K_3^+ C_c^+)}{\partial X} - Da_4 C_b^+ \left(1 + \frac{K_b^+}{\theta^+}\right) C_c^+ - Da_4 K_1^+ C_b^+ \left(1 + \frac{K_b^+}{\theta^+}\right) C_c^+ \\ & - Da_4 K_2^+ C_b^+ C_c^+ - Da_4 C_b^+ K_3^+ C_b^+ C_c^+ - Da_4 C_b^+ K_b^+ K_4^+ C_b^+ \frac{K_b^+}{\theta^+} C_c^+ \quad (38) \end{aligned}$$

$$R_c^+ = \left[1 + \frac{(K_1^+ + K_b^+ K_4^+ C_b^+)}{\theta^+ (1 + K_2^+ + K_3^+ C_b^+)} \right] \quad (39)$$

Note that the hydrodynamic dispersion coefficients of DOM and bacteria are assumed to be the same as that of contaminants. The hydrodynamic dispersion coefficient is composed of the mechanical dispersion coefficient and the Brownian diffusion coefficient, and thus D_1 or D_2 differs from D_c since the Brownian diffusion coefficient of contaminants is not necessarily equal to that of colloidal particles. However, it may be safely assumed that $D_1 = D_2 = D_c$ since the molecular diffusions of contaminants and colloids are negligible compared to the mechanical dispersion (Johnson et al., 1998). In the presence of bacteria and DOM, the total concentration of contaminants in the aqueous phase, C_{ct}^+ , includes the concentrations of contaminants dissolved in the aqueous phase, contaminants sorbed to DOM, and contaminants sorbed to mobile bacteria, and can be expressed as:

$$C_{ct}^+ = C_c^+ (1 + K_2^+ + K_3^+ C_b^+) \quad (40)$$

3. Solution of governing equations

Eqs. (36) and (38) constitute a set of governing equations with two unknowns, C_b^+ and C_c^+ . The analytical solution of Eq. (36) is available under the simplified conditions of constant colloidal concentration, no colloidal sorption to solid matrix, and no bacterial growth and decay (Corapcioglu and Kim, 1995). To obtain the numerical solution and compare with the analytical solution, a fully implicit finite difference method is applied. This method leads to a system of linear algebraic equations with a tridiagonal coefficient matrix, which can be directly solved by the Thomas algorithm (Chapra and Canale, 1998). The numerical solution shows excellent match with the analytical solution (Kim, 2001).

To solve the coupled Eqs. (36) and (38), Eq. (36) is solved for C_b^+ at time step $(n+1)$, where n is the previous time at which all variables including C_c^+ are known. Then, Eq. (38) is solved for C_c^+ at time step $(n+1)$. The following initial and boundary conditions are used in the simulation:

$$C_b^+(X, 0) = C_c^+(X, 0) = 0 \quad (41)$$

$$-\frac{1}{Pe} \frac{\partial C_b^+}{\partial X}(0, T) + C_b^+(0, T) = \begin{cases} 1 & \text{at } 0 < T \leq T_0 \\ 0 & \text{at } T > T_0 \end{cases} \quad (42)$$

$$-\frac{1}{Pe} \frac{\partial C_c^+}{\partial X}(0, T) + C_c^+(0, T) = \begin{cases} 1 & \text{at } 0 < T \leq T_0 \\ 0 & \text{at } T > T_0 \end{cases} \quad (43)$$

$$\frac{\partial C_b^+}{\partial X}(1, T) = \frac{\partial C_c^+}{\partial X}(1, T) = 0 \quad (44)$$

where T_0 is the duration of contaminants and bacteria flux injection.

4. Results and discussion

4.1. Simulation results

The proposed model is used to simulate the transport of hydrophobic organic contaminants in an aquifer where riverbank filtration is practiced. It is assumed in the simulation that bacteria and DOM move through the aquifer simultaneously with the contaminants, and colloidal particles other than bacteria and DOM are absent. In addition, steady-state influent water quality is imposed even though significant seasonal changes of river water quality are observed in real situations, and flow between the river and pumping well is idealized as one-dimensional and horizontal. The idealization of flow condition can be justified due to the use of a set of pumping wells placed along the river in a riverbank filtration operation (Hoehn and Santschi, 1987; Santschi et al., 1987). The model parameters used in the analysis are mainly from Borden and Bedient (1986), Magee et al. (1991), and Jenkins and Lion (1993) (Table 1). In the simulation, contaminant and bacteria are pulse-loaded for 5 T (relative time) unless otherwise specified.

The temporal and spatial variations of the total aqueous phase effluent contaminant concentration (i.e. at a pumping well) in the absence or presence of DOM and bacteria is given in Fig. 3. In the presence of DOM, the breakthrough curve shifts leftward compared to the case of no colloidal presence, and thus the arrival time of relative peak concentration is shortened. In the presence of both DOM and bacteria, the curve shifts slightly further left (Fig. 3a), indicating the enhancement of contaminant transport. When simulation results are presented as relative contaminant concentration along the relative traveling distance (X) (Fig. 3b), the presence of DOM and bacteria results in a shift of the concentration profile toward right relative to that of no colloidal presence, illustrating the increment of contaminant mobility. Simulation results demonstrate that contaminant transport is enhanced markedly in the presence of DOM and bacteria, and the impact of DOM on contaminant mobility is greater than that of bacteria under examined conditions. It should be mentioned that in a certain field situation, DOM transport may be retarded due to its sorption on solid matrix, and thus the degree of contaminant transport facilitation by DOM may be reduced.

Table 1
Model parameters used in numerical simulation

Parameters		Values
n	Porosity	0.36
Pe	Peclet number	50
K_b^+	Relative equilibrium distribution coefficient of bacteria between the aqueous phase and solid matrix	15.31
K_1^+	Relative equilibrium distribution coefficient of contaminant between the aqueous phase and solid matrix	55.56
K_2^+	Relative equilibrium distribution coefficient of contaminant between the aqueous phase and DOM	0.44
K_3^+	Relative equilibrium distribution coefficient of contaminant between the aqueous phase and mobile bacteria	0.22
K_4^+	Relative equilibrium distribution coefficient of contaminant between the aqueous phase and immobile bacteria	0.22
Da_1	Damköhler number for mobile bacteria decay	3.34×10^{-2}
Da_2	Damköhler number for immobile bacteria decay	3.34×10^{-2}
Da_3	Damköhler number for bacteria growth	6.05×10^{-7}
Da_4	Damköhler number for contaminant utilization	1.24×10^{-3}
Da_5	Damköhler number for DOM utilization	9.02×10^{-6}

The temporal variation of the total aqueous phase effluent contaminant concentration in the presence or absence of bacterial deposition on solid matrix is demonstrated in Fig. 4. During the simulation, bacteria are present at all times. Without bacterial deposition on solid matrix, all the bacteria present in the aqueous phase behave like conservative mobile colloids. With no bacterial deposition, the breakthrough curve shifts leftward compared to the case of bacterial deposition, and the arrival time of relative peak concentration is reduced (Fig. 4), showing that the mobility of contaminants increases when no bacterial deposition occurs. The result implies that mobility of contaminants associated with colloids is closely related with mobility of colloids. According to Jenkins and Lion (1993), who observed the bacteria-facilitated transport of phenanthrene, bacterial isolates used in the experiment showed different mobility in porous media, which was expressed in terms of bacterial distribution coefficient on solid matrix (K_{cell}^s). Furthermore, bacterial mobility varies depending on the chemical conditions of the aquifer such as pH and ionic strength (Fontes et al., 1991). Therefore, the degree of bacterium-related contaminant transport in riverbank filtration may be influenced by bacterial type and chemical conditions at a riverbank filtration site.

The extent of contaminant transport enhancement largely depends upon colloidal concentration and its mobility along with its adsorption affinity to contaminants (Roy and Dzombak, 1998). Field studies showed that colloidal concentrations vary widely depending on the conditions of aquatic environments. Regarding water quality, there are diurnal and seasonal changes in most river systems. In summer months, the concentration of colloids such as DOM and bacteria would be in high levels (von Gunten et al., 1991; Bourg and Richard-Raymond, 1994). In groundwater systems, colloidal concentration in undisturbed groundwater is quite low while in the aquifer influenced by surface water its level is high (McCarthy and Degueudre, 1993). When a situation is encountered where

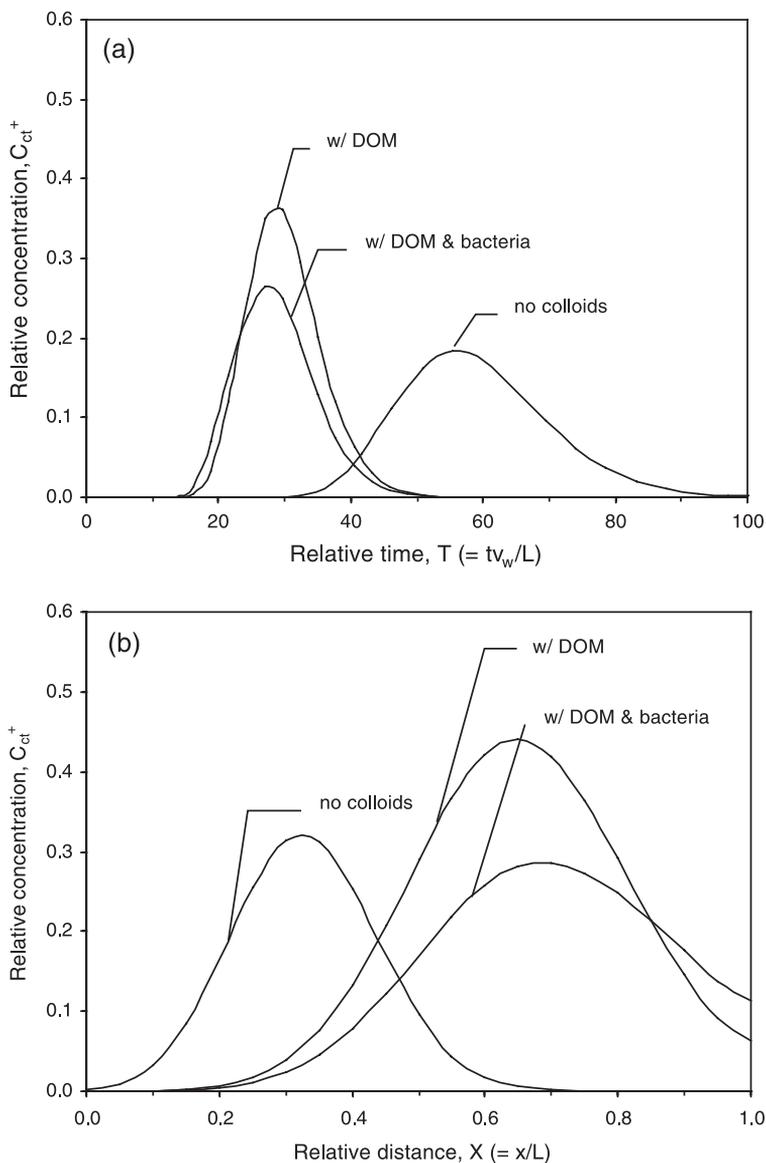


Fig. 3. (a) Temporal variation of total aqueous phase effluent contaminant concentration, C_{ct}^+ and (b) spatial variation of total aqueous phase contaminant concentration, C_{ct}^+ (at $T=20$). Bacterial deposition is taken into account in the simulation.

colloids are present sufficiently in the river and adjacent aquifer, colloid-facilitated contaminant transport is likely to occur in riverbank filtration.

In aquifers, the mobility of colloids depends on solution chemistry as well as the properties of colloids and porous medium. The factors affecting the stability and mobility

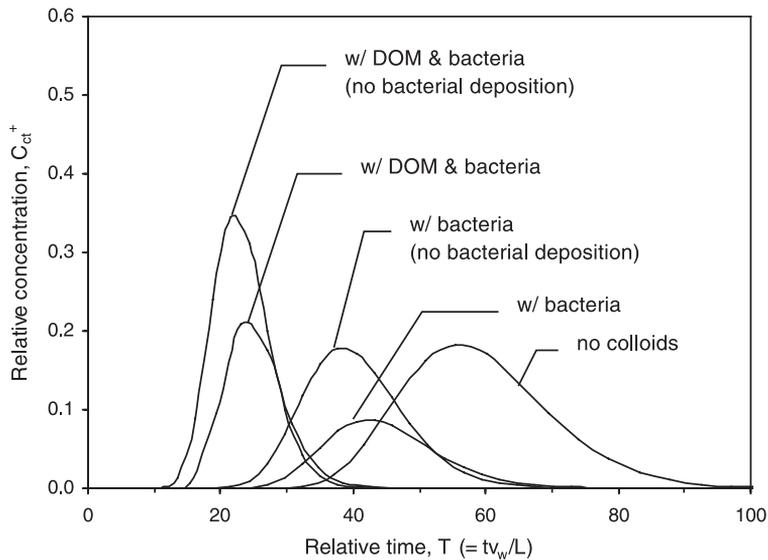


Fig. 4. Temporal variation of total aqueous phase effluent contaminant concentration, C_{ct}^+ , in the presence or absence of bacterial deposition on solid matrix.

of colloids include pH, ionic strength, temperature, and redox potential (Puls and Powell, 1992). High pH, low ionic strength, and low temperature may stabilize colloids, resulting in enhancement of their transport through porous media (McCarthy and Degueudre, 1993). According to Ryan and Gschwend (1990), who observed the colloidal mobilization in groundwater, colloidal concentrations were low in oxic groundwater while anoxic groundwater contained high concentrations of colloids. In river/groundwater system, biological activities related to degradation of organic matter induce considerable variation of water qualities including pH, ionic strength, redox potential, and dissolved metal concentration (von Gunten et al., 1991). The mobility of colloids and their role as contaminant carriers in riverbank filtration were investigated in the River Glatt (Switzerland) and adjacent heterogeneous aquifer (Waber et al., 1987; von Gunten et al., 1988; Waber et al., 1990; Lienert et al., 1994). They found that even if a considerable amount of contaminants (trace metals and radionuclides) was associated with colloids in the river, colloid-related transport of contaminants at a riverbank filtration site was limited. They attributed this phenomenon to coagulation and subsequent mechanical filtration of colloids due to pH change as the river water passed through the aquifer.

In addition, colloidal mobility is closely related to size of colloids and pore-water velocity. According to investigations of Yao et al. (1971) and O'Melia (1980), colloids with a size range from 0.1 to 1.0 μm might be most mobile in a sandy porous medium. For colloidal particles with diameters smaller than 0.1 μm , the removal efficiency increases due to diffusion while for particles larger than 1.0 μm , due to interception and/or sedimentation. Kretzschmar et al. (1997) and Compère et al. (2001) demonstrated that the increment of pore-water velocity decreases the retention of colloids in saturated porous media. Compared to a typical groundwater velocity, flow velocity in the riverbank

filtration site is fast (e.g. 3–6 m day⁻¹ in the River Glatt, Switzerland; 3–5 m day⁻¹ in the Lot River, France) and varies depending on hydraulic conditions (Hoehn and von Gunten, 1989; Bourg and Bertin, 1993).

4.2. Sensitivity analysis

Sensitivity analysis is performed to observe the effect of model parameters on the model behavior. During the sensitivity analysis for a particular parameter, other parameter values remain unchanged. Among the model parameters, results of the following parameter are selectively presented: relative contaminant distribution coefficient between the aqueous phase and solid matrix (K_1^+); relative contaminant distribution coefficient between the aqueous phase and DOM (K_2^+); relative contaminant distribution coefficient between the aqueous phase and bacteria (K_3^+). In the analysis, the range of distribution coefficients is dependent upon the characteristics of colloidal surfaces and solid matrix. In case of Damköhler numbers related to bacterial growth, decay, and contaminant utilization, our model shows no significant variation by one order of magnitude change in their values (data not shown).

The effect of relative contaminant distribution coefficient between the aqueous phase and solid matrix, K_1^+ , on the temporal variation of the total aqueous phase effluent contaminant concentration is illustrated in Fig. 5. The result demonstrates that one order of magnitude change in K_1^+ markedly changes the breakthrough curve. As K_1^+ increases, the breakthrough curve flattens and shifts rightward, and the arrival time of relative peak concentration increases. The lumped parameter K_1^+ represents a total affinity of stationary

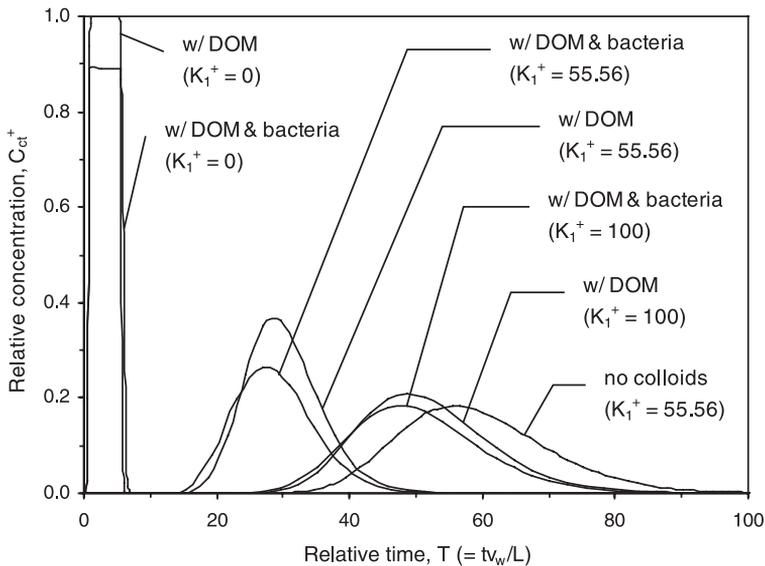


Fig. 5. Effect of contaminant distribution coefficient between the aqueous phase and solid matrix, K_1^+ , on the temporal variation of total aqueous phase effluent contaminant concentration, C_{ct}^+ .

solid matrix to contaminants. As presented in Eq. (34), it is comprised of K_1 (distribution coefficient of contaminants on solid matrix), equivalent to the affinity of solid matrix to contaminants, and ρ_s/θ , equivalent to the concentration of stationary solid phase (i.e. mass of solid matrix per unit volume of water). The K_1 is composed of K_{oc} , the organic carbon/water partitioning coefficient, and f_{oc} , the organic carbon content of solid matrix (i.e. $K_1 = K_{oc}f_{oc}$) (Worch et al., 2002). The K_{oc} is contaminant-specific. Phenanthrene used as the model compound has $K_{oc} = 2.3 \times 10^4 \text{ ml g}^{-1}$ while the K_{oc} of benzo[a]pyrene is 100-fold greater and that of naphthalene is 20-fold less (Fetter, 1999). The f_{oc} is site-specific and varies with field sites (Schwarzenbach and Westall, 1981). Therefore, the affinity of stationary solid matrix to hydrophobic organic contaminants (K_1) depends on the chemical characteristics of contaminants and properties of porous media.

The effect of relative contaminant distribution coefficient between the aqueous phase and DOM, K_2^+ , on the temporal variation of the total aqueous phase effluent contaminant concentration is shown in Fig. 6. The result shows that the model is sensitive to the change of K_2^+ . As K_2^+ increases, the breakthrough curve shifts leftward, and the arrival time of relative peak concentration decreases. The lumped parameter K_2^+ denotes a total affinity of DOM to contaminants and is composed of K_2 (distribution coefficient of contaminants on DOM), equivalent to the affinity of DOM to contaminants, and C_d , equivalent to the concentration of DOM. The K_2 is also related to K_{oc} , the contaminant-specific factor. At riverbank filtration sites, the degree of DOM removal is influenced by hydrological conditions, solution chemistry, site hydrogeology, and well placement. According to the study of Čosović et al. (1996), who observed the changes of organic matter in the river-aquifer (small stream-alluvial aquifer) system (Croatia), after a rain event the concentration

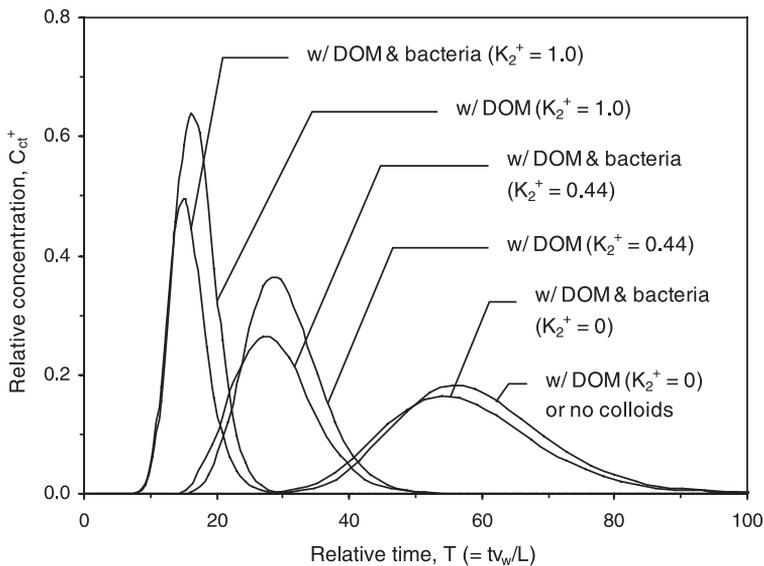


Fig. 6. Effect of contaminant distribution coefficient between the aqueous phase and DOM, K_2^+ , on the temporal variation of total aqueous phase effluent contaminant concentration, C_{ct}^+ .

of DOM in the adjacent aquifer increased significantly due to an increase of surface water infiltration. Thus, concentration of DOM involved in the enhancement of contaminant transport (e.g. humic substance) may change in the aquifer during the riverbank filtration operation. From the observation of pyrene sorption to humic substance, [Chin et al. \(1997\)](#) reported that binding capacity of organic matter was positively related to the aromatic content of organic matter. Therefore, total affinity of DOM to hydrophobic organic contaminants (K_2^+) depends on the chemical properties of both contaminants and DOM as well as on the concentration of DOM that can play the role of a contaminant carrier.

Inorganic colloids such as clay minerals, iron oxides, and kaolinite may be present in the river and groundwater, though not considered in this study, affecting the transport behavior of contaminants in riverbank filtration. From a field study in two creosote-contaminated aquifers (Denmark), [Villholth \(1999\)](#) found that the partitioning of PAHs to inorganic colloids occurred at the site where the mass fraction of organic matter associated with colloids is high and observed that only larger colloids (>100 nm) were involved with the partitioning of PAHs. As observed by several researchers ([Amirbahman and Olson, 1993](#); [Kretzschmar et al., 1995](#); [Deshiikan et al., 1998](#)), organic matter coatings could alter the surface properties of inorganic colloids, making them stable in groundwater, preventing their coagulation and filtration, and consequently traveling over significance distances while carrying contaminants on them. [Gschwend and Reynolds \(1987\)](#) have observed the formation and mobility of monodisperse ferrous phosphate colloids in the Cape Cod aquifer (Massachusetts, USA), which was contaminated by infiltration of secondary treated sewage, indicating that the colloids with organic matter coatings had a high potential as contaminant carriers due to their mobility.

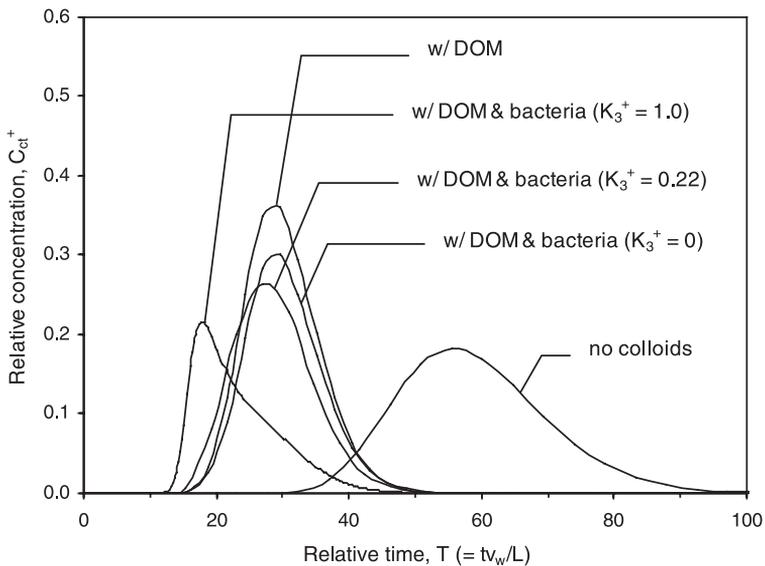


Fig. 7. Effect of contaminant distribution coefficient between the aqueous phase and the surfaces of mobile bacteria, K_3^+ , on the temporal variation of total aqueous phase effluent contaminant concentration, C_{ct}^+ .

Table 2

Summary of simulation result and illustration of effect of the ratio of $K_1^+/K_2^+/K_3^+$ on the contaminant breakthrough curves

K_1^+	K_2^+	K_3^+	$K_1^+/K_2^+/K_3^+$	Arrival time of peak concentration	Relative peak concentration
55.56	0.44	0.22	253:2:1	27.0	0.26
100	0.44	0.22	455:2:1	48.0	0.18
55.56	1	0.22	253:4.5:1	15.0	0.49
55.56	0.44	1	56:0.4:1	18.3	0.21

The effect of relative contaminant distribution coefficient between the aqueous phase and mobile bacteria, K_3^+ , on the temporal variation of the total aqueous phase effluent contaminant concentration is presented in Fig. 7. It is assumed that K_3^+ is equal to K_4^+ . The result illustrates that the model is sensitive to the change of K_3^+ . With an increase of K_3^+ , the breakthrough curve shifts leftward, and the arrival time of relative peak concentration decreases. The lumped parameter K_3^+ indicates a total affinity of bacteria to contaminants and is composed of K_3 (distribution coefficient of contaminants on bacteria), equivalent to the affinity of bacteria to contaminants, and C_{b0} , equivalent to the concentration of bacteria. Jenkins and Lion (1993) showed that the bacterial isolates used in the experiment showed different affinity to phenanthrene, expressed in terms of contaminant distribution coefficient on bacteria (K_d^{cell}). In aquatic environments, a community of bacteria would include species with highly different shapes, sizes, and surface properties that would experience differential interaction with both contaminants and surfaces of porous media. In riverbank filtration, the degree of bacteria-associated contaminant transport depends on types of bacteria and their concentrations.

In the model system (four-phase system), the affinities of stationary solid matrix, DOM, and bacteria to contaminants are described by K_1^+ , K_2^+ , and K_3^+ , respectively. As illustrated in Table 2, the magnitude of contaminant transport enhancement in the four-phase system is closely related to the ratio of $K_1^+/K_2^+/K_3^+$. The increase of K_1^+ , which indicates the increment of affinity of solid matrix (immobile phase) to contaminants relative to DOM and bacteria (mobile phases), results in the reduction of contaminant mobility. On the contrary, the increase of K_2^+ or K_3^+ causes the enhancement of contaminant mobility.

5. Summary and conclusions

In this study with a mathematical model developed using an equilibrium-based approach, the transport of hydrophobic organic contaminants in the presence of DOM and bacteria is investigated in the aquifer where riverbank filtration is practiced. The simulation results show that the contaminant mobility in riverbank filtration might increase significantly due to the reduction of retardation by colloid-mediated contaminant transport. The sensitivity analysis illustrates that contaminant breakthrough curves are very sensitive to the changes of various contaminant-related distribution coefficients. In the model system, the degree of colloid-associated contaminant transport depends on the ratio of three relative contaminant distribution coefficients (K_1^+ , K_2^+ , and K_3^+), which quantifies the

distribution of contaminants in the four-phase system. In riverbank filtration, migration of colloid-associated contaminants may vary depending on the biochemical and hydro-geologic conditions at the operation site. Especially in storm events when loads of contaminants and colloids are high and the infiltration rate increases, colloid-associated contaminants are likely to break through the aquifer and appear at the production wells in riverbank filtration. Therefore, phenomenon of colloid-related contaminant transport must be accounted during the riverbank filtration operation. Though developed based on the simplistic representation of riverbank filtration conditions, this theoretical model could help to understand the transport behavior of the hydrophobic organic contaminants during the riverbank filtration operation.

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References

- Ahel, M., Schaffner, C., Giger, W., 1996. Behaviour of alkylphenol polyethoxylate surfactants in the aquatic environment: III. Occurrence and elimination of their persistent metabolites during infiltration of river water to groundwater. *Water Res.* 30 (1), 37–46.
- Amirbahman, A., Olson, T.M., 1993. Transport of humic matter-coated hematite in packed beds. *Environ. Sci. Technol.* 27 (13), 1993.
- Bellin, C.A., Rao, P.S.C., 1993. Impact of bacterial biomass on contaminant sorption and transport in a subsurface soil. *Appl. Environ. Microbiol.* 59 (6), 1813–1820.
- Borden, R.C., Bedient, P.B., 1986. Transport of dissolved hydrocarbons influenced by oxygen-limited biodegradation: 1. Theoretical development. *Water Resour. Res.* 22 (13), 1973–1982.
- Bourg, A.C.M., Bertin, C., 1993. Biogeochemical processes during the infiltration of river water into an alluvial aquifer. *Environ. Sci. Technol.* 27 (4), 661–666.
- Bourg, A.C.M., Bertin, C., 1994. Seasonal and spatial trends in manganese solubility in an alluvial aquifer. *Environ. Sci. Technol.* 28 (5), 868–876.
- Bourg, A.C.M., Richard-Raymond, F., 1994. Spatial and temporal variability in the water redox chemistry of the M27 experimental site in the Drac River calcareous alluvial aquifer (Grenoble, France). *J. Contam. Hydrol.* 15, 93–105.
- Brusseau, M.L., Rao, P.S.C., 1989. Sorption nonideality during organic contaminant transport in porous media. *CRC Crit. Rev. Environ. Control* 19 (1), 33–99.
- Chapra, S.C., Canale, R.P., 1998. *Numerical Methods for Engineers: with Programming and Software Applications*, 3rd ed. McGraw-Hill, Boston, MA, USA.
- Chin, Y.-P., Aiken, G.R., Danielsen, K.M., 1997. Binding of pyrene to aquatic and commercial humic substances: the role of molecular weight and aromaticity. *Environ. Sci. Technol.* 31 (6), 1630–1635.
- Compère, F., Porel, G., Delay, F., 2001. Transport and retention of clay particles in saturated porous media: influence of ionic strength and pore velocity. *J. Contam. Hydrol.* 49, 1–21.
- Corapcioglu, M.Y., Jiang, S., 1993. Colloid-facilitated groundwater contaminant transport. *Water Resour. Res.* 29 (7), 2215–2226.
- Corapcioglu, M.Y., Kim, S., 1995. Modeling facilitated contaminant transport by mobile bacteria. *Water Resour. Res.* 31 (11), 2639–2647.
- Ćosović, B., Hršak, D., Vojvodić, V., Krznarić, D., 1996. Transformation of organic matter and bank filtration from a polluted stream. *Water Res.* 30 (12), 2921–2928.

- Deshiikan, S.R., Eschenazi, E., Papadopoulos, K.D., 1998. Transport of colloids through porous beds in the presence of natural organic matter. *Colloids Surf., A Physicochem. Eng. Asp.* 145, 93–100.
- Doussan, C., Poitevin, G., Ledoux, E., Detay, M., 1997. Riverbank filtration: modeling of the changes in water chemistry with emphasis on nitrogen species. *J. Contam. Hydrol.* 25, 129–156.
- Enfield, C.G., Bengtsson, G., Lindqvist, R., 1989. Influence of macromolecules on chemical transport. *Environ. Sci. Technol.* 23 (10), 1278–1286.
- Fetter, C.W., 1999. *Contaminant Hydrogeology*, 2nd ed. Prentice-Hall, Upper Saddle River, NJ, USA.
- Fontes, D.E., Mills, A.L., Hornberger, G.M., Hermann, J.S., 1991. Physical and chemical factors influencing transport of microorganisms through porous media. *Appl. Environ. Microbiol.* 57 (9), 2473–2481.
- Frimmel, F.H., 1998. Characterization of natural organic matter as major constituents in aquatic systems. *J. Contam. Hydrol.* 35, 201–216.
- Grolimund, D., Borkovec, M., Barmettler, K., Sticher, H., 1996. Colloid-facilitated transport of strongly sorbing contaminants in natural porous media: a laboratory column study. *Environ. Sci. Technol.* 30 (10), 3118–3123.
- Gschwend, P.M., Reynolds, M.D., 1987. Monodisperse ferrous phosphate colloids in an anoxic groundwater plume. *J. Contam. Hydrol.* 1, 309–327.
- Hoehn, E., Santschi, P.H., 1987. Interpretation of tracer displacement during infiltration of river water to groundwater. *Water Resour. Res.* 23 (4), 633–640.
- Hoehn, E., von Gunten, H.R., 1989. Radon in groundwater: a tool to assess infiltration from surface waters to aquifers. *Water Resour. Res.* 25 (8), 1795–1803.
- Jenkins, M.B., Lion, L.W., 1993. Mobile bacteria and transport of polynuclear aromatic hydrocarbons in porous media. *Appl. Environ. Microbiol.* 59 (10), 3306–3313.
- Johnson, W.P., Cabral, K., Lion, L.W., Corapcioglu, M.Y., 1998. Reconciliation of expressions for the modified retardation factor and incorporation of non-linear contaminant sorption to the stationary phase. *J. Contam. Hydrol.* 32, 247–266.
- Kim, S.B., 2001. Modeling transport of colloid-facilitated contaminants and *Cryptosporidium parvum* oocysts in riverbank filtration. PhD dissertation, Texas A&M Univ., College Station, TX, USA.
- Kim, S., Corapcioglu, M.Y., 1996. A kinetic approach to modeling mobile bacteria-facilitated groundwater contaminant transport. *Water Resour. Res.* 32 (2), 321–331.
- Kretzschmar, R., Robarge, W.P., Amoozegar, A., 1995. Influence of natural organic matter on colloid transport through saprolite. *Water Resour. Res.* 31 (3), 435–445.
- Kretzschmar, R., Barmettler, K., Grolimund, D., Yan, Y.-D., Borkovec, M., Sticher, H., 1997. Experimental determination of colloid deposition rates and collision efficiencies in natural porous media. *Water Resour. Res.* 33 (5), 1129–1137.
- Kuehn, W., Mueller, U., 2000. Riverbank filtration: an overview. *J. AWWA* 92 (12), 60–69.
- Kuhn, E.P., Colberg, P.J., Schnoor, J.L., Wanner, O., Zehnder, A.J.B., Schwarzenbach, R.P., 1985. Microbial transformations of substituted benzenes during infiltration of river water to groundwater: laboratory column studies. *Environ. Sci. Technol.* 19 (10), 961–968.
- Lienert, C., Short, S.A., von Gunten, H.R., 1994. Uranium infiltration from a river to shallow groundwater. *Geochim. Cosmochim. Acta* 58 (24), 5455–5463.
- Lindqvist, R., Enfield, C.G., 1992. Biosorption of dichlorodiphenyltrichloroethane and hexachlorobenzene in groundwater and its implications for facilitated transport. *Appl. Environ. Microbiol.* 58 (7), 2211–2218.
- Literathy, P., László, F., 1996. Processes affecting the quality of bank-filtered water. In: Kivimäki, A.-L., Suokko, T. (Eds.), *Proceedings of the International Symposium on Artificial Recharge of Groundwater*, NHP Report No. 38. Nordic Hydrological Programme, Helsinki, Finland, pp. 53–64.
- Liu, H., Amy, G., 1993. Modeling partitioning and transport interactions between natural organic matter and polynuclear aromatic hydrocarbons in groundwater. *Environ. Sci. Technol.* 27 (8), 1553–1562.
- Magee, B.R., Lion, L.W., Lemley, A.T., 1991. Transport of dissolved organic macromolecules and their effect on the transport of phenanthrene in porous media. *Environ. Sci. Technol.* 25 (2), 323–331.
- Marmonier, P., Fontieille, D., Gibert, J., Vanek, V., 1995. Distribution of dissolved organic carbon and bacteria at the interface between the Rhône River and its alluvial aquifer. *J. North Am. Benthol. Soc.* 14 (3), 382–392.
- Matthess, G., Pekdeger, A., Schroeter, J., 1988. Persistence and transport of bacteria and viruses in groundwater—a conceptual evaluation. *J. Contam. Hydrol.* 2, 171–188.

- McCarthy, J.F., Degueudre, C., 1993. In: Buffle, J., van Leeuwen, H.P. (Eds.), *Environmental Particles*, vol. 2. Lewis Publishers, Boca Raton, FL, USA. Chap. 6.
- McCarthy, J.F., Gu, B., Liang, L., Mas-Pla, J., Williams, T.M., Yeh, T.-C.J., 1996. Field tracer tests on the mobility of natural organic matter in a sandy aquifer. *Water Resour. Res.* 32 (5), 1223–1238.
- Mills, W.B., Liu, S., Fong, F.K., 1991. Literature review and model (COMET) for colloid/metals transport in porous media. *Ground Water* 29 (2), 199–208.
- Noell, A.L., Thompson, J.L., Corapcioglu, M.Y., Triay, I.R., 1998. The role of silica colloids on facilitated cesium transport through glass bead columns and modeling. *J. Contam. Hydrol.* 31, 23–56.
- O'Melia, C.R., 1980. Aquasols: the behavior of small particles in aquatic systems. *Environ. Sci. Technol.* 14, 1052–1060.
- Piet, G.J., Zoeteman, B.C.J., 1980. Organic water quality changes during sand bank and dune filtration of surface waters in the Netherlands. *J. AWWA* 72 (7), 400–404.
- Puls, R.W., Powell, R.M., 1992. Transport of inorganic colloids through natural aquifer material: implications for contaminant transport. *Environ. Sci. Technol.* 26 (3), 614–621.
- Ray, C., Soong, T.W., Lian, Y.Q., Roadcap, G.S., 2002. Effect of flood-induced chemical load on filtrate quality at bank filtration sites. *J. Hydrol.* 266, 235–258.
- Roy, S.B., Dzombak, D.A., 1998. Sorption nonequilibrium effects on colloid-enhanced transport of hydrophobic organic compounds in porous media. *J. Contam. Hydrol.* 30, 179–200.
- Ryan, J.N., Gschwend, P.M., 1990. Colloid mobilization in two Atlantic coastal plain aquifers: field studies. *Water Resour. Res.* 26 (2), 307–322.
- Saiers, J.E., Hornberger, G.M., 1996. Modeling bacteria-facilitated transport of DDT. *Water Resour. Res.* 32 (5), 1455–1459.
- Santschi, P.H., Hoehn, E., Lueck, A., Farrenkothen, K., 1987. Tritium as a tracer for the movement of surface water and groundwater in the Glatt Valley, Switzerland. *Environ. Sci. Technol.* 21 (9), 909–916.
- Schellenberg, K., Leuenberger, C., Schwarzenbach, R.P., 1984. Sorption of chlorinated phenols by natural sediments and aquifer materials. *Environ. Sci. Technol.* 18 (9), 652–657.
- Schwarzenbach, R.P., Westall, J., 1981. Transport of nonpolar organic compounds from surface water to groundwater. Laboratory sorption studies. *Environ. Sci. Technol.* 15 (11), 1360–1367.
- Schwarzenbach, R.P., Giger, W., Hoehn, E., Schneider, J.K., 1983. Behavior of organic compounds during infiltration of river water to groundwater: field studies. *Environ. Sci. Technol.* 17 (8), 472–479.
- Sontheimer, H., 1980. Experience with riverbank filtration along the Rhine River. *J. AWWA* 72 (7), 386–390.
- Tsezos, M., Seto, W., 1986. The adsorption of chloroethanes by microbial biomass. *Water Res.* 20 (7), 851–858.
- van der Kooij, D., Groennou, J.T., Kruithof, J.C., Noordam, P.C., Noordsij, A., Stuyfzand, P.J., van der Gaag, M.A., 1985. Water quality aspects of river bank filtration in the Netherlands. *Water Supply* 3, 41–50.
- Villholth, K.G., 1999. Colloid characterization and colloidal phase partitioning of polycyclic aromatic hydrocarbons in two creosote-contaminated aquifers in Denmark. *Environ. Sci. Technol.* 33 (5), 691–699.
- von Gunten, H.R., Waber, U.E., Krähenbühl, U., 1988. The reactor accident at chernobyl: a possibility to test colloid-controlled transport of radionuclides in a shallow aquifer. *J. Contam. Hydrol.* 2, 237–247.
- von Gunten, H.R., Karametaxas, G., Krähenbühl, U., Kuslys, M., Giovanoli, R., Hoehn, E., Keil, R., 1991. Seasonal biogeochemical cycles in riverborne groundwater. *Geochim. Cosmochim. Acta* 55, 3597–3609.
- Waber, U.E., von Gunten, H.R., Krähenbühl, U., 1987. The impact of the Chernobyl accident on a river/groundwater aquifer. *Radiochim. Acta* 41, 191–198.
- Waber, U.E., Lienert, C., von Gunten, H.R., 1990. Colloid-related infiltration of trace metals from a river to shallow groundwater. *J. Contam. Hydrol.* 2, 251–265.
- Worch, E., Grischek, T., Börnick, H., Eppinger, P., 2002. Laboratory tests for simulating attenuation processes of aromatic amines in riverbank filtration. *J. Hydrol.* 266, 259–268.
- Yao, K.M., Habibian, M.T., O'Melia, C.R., 1971. Water and wastewater filtration: concepts and applications. *Environ. Sci. Technol.* 5 (11), 1105–1112.