Microbial Removal Rates in Subsurface Media Estimated From Published Studies of Field Experiments and Large Intact Soil Cores

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Information about the microbial removal efficiencies of subsurface media is essential for assessing the risk of water contamination, estimating setback distances between disposal fields and receiving waters, and selecting suitable sites for wastewater reclamation. By analyzing published data from field experiments and large intact soil cores, an extensive database of microbial removal rates was established for a wide range of subsurface media. High microbial removal rates were found in volcanic soils, pumice sand, fine sand, and highly weathered aquifer rocks. Low removal rates were found in structured clayey soils, stony soils, coarse gravel aquifers, fractured rocks, and karst limestones. Removal rates were lower for enteroviruses than for other human viruses; for MS2 phage than for other phage species; for waste-associated microbes than for those cultivated in the laboratory; and for contaminated media than for uncontaminated media. Microbial removal rates are inversely correlated with infiltration rates and transport velocity. The assumption of first-order law, or a constant removal rate (when the transport scale reaches a representative elementary volume), is appropriate for most of field data analyzed. However 30% of the datasets (26 out of 87 pairs) are better described with the power law, implying reduced removal rates with transport distance. The latter is most prominent for organically contaminated media, especially in relatively fine aquifer media. The presence of organic matter, heterogeneity in microbial properties, change in solution chemistry, detachment, and physical straining, may have caused the discrepancies from the first-order law traditionally used in transport models for describing microbial removal.

Published in J. Environ. Qual. 38:1531–1559 (2009).
doi:10.2134/jeq2008.0379
Received 21 Aug. 2008.
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Many waterborne disease outbreaks are caused by the consumption of groundwater contaminated by pathogens (Beller et al., 1997; Craun et al., 2002; Fong et al., 2007; Miettinen et al., 2001; Parshionikar et al., 2003). Recent studies have demonstrated that not only bacteria, but also enteroviruses are widespread in groundwater (Abbaszadegan et al., 2003; Borchardt et al., 2003, 2004, 2007; Fout et al., 2003). Land disposal of human and animal effluent and sludge is a major source of pathogens in groundwater systems.

Although subsurface media act as natural filters and buffers that can mitigate microbial contamination, they vary widely in their ability to remove microbial contaminants. To provide accurate evaluations of the risk of microbial contamination of waters under effluent land disposal, to establish safe setback distances between receiving waters and disposal fields, and to select suitable sites for wastewater reclamation, the ability of subsurface media in microbial attenuation must be evaluated and parameterized. Such information will be very helpful for improving resource management and monitoring of groundwater contamination.

Field study data are the most relevant and reliable for resource management. However, comparatively less information is available on microbial transport from field studies than from laboratory column studies. Laboratory column studies are often not representative of field conditions, because the physical-chemical properties associated with media heterogeneity and transport scale greatly influences microbial transport. Media heterogeneity and transport scale are difficult to replicate in the laboratory and repacking generally reduces the macropore structure of the subsurface media. Microbial removal determined from laboratory column studies can be one to three orders of magnitude greater than that determined from field conditions (Table 1). Therefore if setback distances are estimated using laboratory-derived results, they can be underestimated by orders of magnitude. As setback distance estimations are extremely sensitive to microbial removal rates (Pang et al., 2003; Schijven et al., 2006; Yates and Jury, 1995), being exponentially correlated (Pang et al., 2003), the accuracy of removal rates for the problems of interest is critical.

Although some studies have derived setback distances for certain aquifers (Masciopinto et al., 2008; Pang et al., 2003, 2005; Schijven et al., 2006; van der Wielen et al., 2008; Yates and Yates, 1989), estimations of these distances only consider microbial transport through

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Abbreviations: BTC, breakthrough curve; C_p/C_o, peak concentration observed down gradient relative to its injection concentration; REV, representative elementary volume.

Table 1. Comparison of microbial removal observed under field and laboratory conditions.

References	Media	Microbe	Condition	Log_{10} reduction	*x* (m)	Log/m
Medema et al. (2000)	Coarse and fine gravel with sand, Roosteren	F-RNA phages and coliforms	Field	4.00	15.00	0.27
Hijnen et al. (2005)	From same site as in Medema et al. (2000)	*E. coli*	Column	4.10–4.80	0.50	8.2–9.6
		MS2 phages	Column	1.30–3.40	0.50	2.6–6.8
Schijven et al. (1999)	Dune sand aquifer, Castricum	MS2 phages	Field	3.30	3.80	0.87
Hijnen et al. (2005)	From same site as in Schijven et al. (1999)	MS2 phages	Column	2.20–3.30	0.50	4.4–6.6
Harvey et al. (1995)	Sand and fine gravel aquifer, Cape Code	Protozoa	Field	2.92	3.60	0.81
	Taken from the same site	Protozoa	Column	2.92	0.60	4.87
DeFlaun et al. (1997)	Fine-medium sand	Bacteria	Field	2.00	0.50	4.00
	Taken from the same site	Bacteria	Column	4.70	0.02	235
Harvey et al. (2008)	Karst limestone	2.9 μm microspheres	Field	1.40	97.00	0.01
	Taken from the same site	2.9 μm microspheres	Column	1.40	0.17	8.47
Harvey et al. (2002)	Sand (grain size 0.5–1 mm) from Cape Cod	*Spumella guttula*-DAPI	Field	0.39–0.72	1–3.6	0.18–0.39
	Materials used in column experiments were	*Spumella guttula*-DAPI	Column	0.41–1.10	0.60	0.68–1.83
	taken from the same site.	*Spumella guttula*-HE	Field	0.89–2.0	1–3.6	0.56–0.89
		Spumella guttula-HE	Column	3.00	0.60	5.00
Smith et al. (1985)	Soil	*E. coli*	Intact core	2 log lower than disturbed core		
	Taken from the same site	*E. coli*	Disturbed core			

saturated zones and they do not take into account microbial reduction through soils and vadose zones (the zones between soils and water tables), due to a lack of information for these media. Thus they are only applicable for a worst-case scenario when the water tables are close to the bottom of disposal systems. Usually, there are soils and vadose zones above the water tables and depending on their thicknesses, the horizontal setback distances required could be significantly reduced. Although some field studies on microbial transport through soils and vadose zones are reported in the literature, most studies do not give removal rates directly, making information transfer difficult.

This study has established an extensive database on the removal rates of a limited number of human pathogenic viruses, bacteriophages, a few groups of bacteria, spores, and protozoa in a wide range of subsurface media under various field conditions, and is a response to demands from many users of scientific information (regulators, environmental managers, utilities, consultants, private organisations, researchers, and academics). The database was accomplished by analyzing a large body of published data obtained from field experiments and large undisturbed soil lysimeters. The magnitudes and distribution patterns of removal rates for microbial transport in subsurface media were identified and summarized. The removal rates provided in this paper are presented in a simple form that can be easily adopted by others for addressing environmental management problems, for example, estimation of setback distances.

Materials and Methods

Basic Concepts

Many factors and processes affect microbial removal rates in subsurface media. These include physical-chemical properties of subsurface media, properties of microbial contaminants, solution chemistry, inactivation, physical straining, the air–water interface in unsaturated media, heterogeneity and preferential flow paths, different flow mechanisms for karsts and fractured media from those for porous media, source/carrier characteristics of the microbial contaminants (e.g., continuous or intermittent loading, and loading duration), and management practices (e.g., injection vs. surface application, etc.). It will be superfluous to directly consider these factors and processes in the methods to be used for analyzing a large amount of literature data, which would increase the complexity of data analysis.

For this study, a simplified integrative and straightforward approach was sought so that comparable removal rates could be derived for various subsurface media. Therefore, the following assumptions and simplifications were made: microbial removal in both saturated and unsaturated zones is a first-order irreversible process, and microbial transport is at a steady state, and predominantly occurs along one-dimensional preferential flow paths by advection with negligible dispersion. On the basis of these assumptions, the following concepts were employed.

In conventional transport models, microbial removal is considered to be a first-order process:

$$\frac{dC}{dt} = -kC \qquad [1]$$

where C is the microbial concentration in solution (M/L^3), k is the first-order temporal removal rate (T^{-1}), and t is the time (T).

For a constant velocity $\frac{dx}{dt} = V$ along the flow direction, where x is the distance traveled (L) and V is the average pore-water velocity of a microbial tracer (L/T), Eq. [1] becomes

$$\frac{dC}{dt} = \frac{dC}{dx}\frac{dx}{dt} = \frac{dC}{dx}V = -kC \qquad [2]$$

$$\text{Replacing } \lambda = \frac{k}{V} \qquad [3]$$

Equation [2] becomes

$$\frac{dC}{dx} = -\lambda C \qquad [4]$$

where λ is the spatial removal rate of the microbial tracer (L^{-1}). Equation [4] implies that microbial concentration decreases exponentially with travel distance as a first-order process.

It should be noted that the constant removal rate defined in the above first-order process should be considered as a large-scale average. It is assumed here that the concept of representative elementary volume, REV, could be applied to removal rates. The REV is the smallest sample size that has properties representative of the media mass, that is at this sample size the parameter of interest is scale invariant (Bear, 1972). It is assumed that, at a transport scale large enough, the effects of heterogeneities (in geochemical and physical properties of the porous media, solution chemistry, microbial populations, strains and isolates, etc.) will become not so important and that the removal rate can be treated as a constant. The REV assumption about the removal rate will be validated using field experimental data later on in this article.

Although Eq. [4] has the same formula as the filtration theory (Iwasaki, 1937; Yao et al., 1971; Logan et al., 1995), the first-order rate assigned here is effectively a removal rate that lumps the effects of all irreversible processes (irreversible attachment, inactivation, straining, and for unsaturated media also air-water interaction) assuming all irreversible processes follow a first-order law. Converting from the natural log in the original form to $\log_{10}$ by multiplying by a factor of 2.3, λ measures the relative log-reduction in microbial concentration achieved per unit of distance traveled.

Under steady-state conditions, the solution of Eq. [4] for a continuous solution input is given by (Matthess et al., 1988)

$$\lambda = -\frac{\ln\left(\frac{C_p}{C_o}\right)}{x} = -2.30\frac{\log_{10}\left(\frac{C_p}{C_o}\right)}{x} \quad [5]$$

in which, C_p is the effluent concentration (M/L^3) at the plateau (i.e., peak) of the breakthrough curve (BTC), and C_0 is the influent concentration (M/L^3). Therefore the λ value can be interpreted from the slope of a $\log_{10}$ (C_p/C_0) vs. x plot or concentration reduction measured at a single distance x.

Experiments with pulse inputs are often preferable to those with continuous inputs as a lower volume of solution is needed. Kretzschmar et al. (1997) modified Eq. [5] for a pulse input of solution

$$\lambda = -\frac{\ln\left(\frac{Q}{N_0}\int_0^{t_f} C(t)dt\right)}{x} = -2.30\frac{\log_{10}\left(\frac{Q}{N_0}\int_0^{t_f} C(t)dt\right)}{x} \quad [6]$$

where Q is the flow rate (L^3/T), N_0 is the total amount of the microbial tracer injected (M), t_f is the time at which the solution pulse has completely moved through the column (T). The term in brackets in Eq. [6] corresponds to the mass recovery of the microbial tracer (i.e., the zero moment), which can be obtained by integrating the entire BTC and normalizing it to the total amount of the microbial tracer injected.

Although the above equations are derived from 1-D transport, they are valid for 3-D transport as long as the velocity is constant and is aligned in the x-direction. Likewise they are also valid for unsaturated flow at a steady state.

Alternatively, if BTC data are available, λ can be converted from k, which can be determined by fitting the experimental BTC with the convection-dispersion transport equation that considers first-order removal

$$\frac{dC}{dt} = D\frac{d^2C}{dx^2} - V\frac{dC}{dx} - kC \quad [7]$$

where D is the dispersion coefficient for the microbial tracer (L^2/T). Equation [7] is applicable for both saturated and unsaturated conditions under a steady state.

Experimental Evaluation of Different Methods

The methods described above, based on peak-concentration, mass balance, and curve fitting, have different levels of complexity and provide alternative approaches to estimate removal rates, depending on the type of data available (either concentration vs. location or concentration vs. time), and the source inputs (continuous or pulse input). Two of these methods require concentration BTC data. However, most field studies only produce sparse concentration vs. distance data. Two questions arise: Can the peak-concentration method for continuous inputs be used to approximate pulse inputs when BTC data are not available? How do the results differ for the same dataset when it is derived from curve-fitting of a transport model or the simple mass balance method?

To answer these questions, the experimental data obtained from undisturbed large soil lysimeters presented in Pang et al. (2008) were analyzed, selecting those with complete BTCs. CXTFIT, version 2 (Toride et al., 1995), developed to simulate contaminant transport under steady flow for both unsaturated and saturated conditions, was used for curve-fitting. As microbial particles and solutes travel through different pathways due to size-exclusion, only the BTC data of the microbial tracer were needed and parameters v, D and k were optimized. Results of CXTFIT curve fitting are listed in Table 2. As microbial removal in these soils was essentially irreversible (Pang et al., 2008), the removal rates, k, estimated from the one-region CXTFIT model are similar to those k_{mim} estimated by Pang et al. (2008) using the two-region mobile-immobile water model of HYDRUS-1D (Table 2). The model-derived temporal removal rate k is then converted to the spatial removal rate λ using Eq. [3].

These data are further analyzed using Eq. [5] and [6], and the results are compared (Table 2). Table 2 shows that the removal rates estimated using the three methods are comparable and are mostly similar. As expected, most removal rates (61%) determined from peak concentrations are slightly higher than those determined from mass balance because only the highest concentrations are considered in the estimations. The results from CXTFIT model are 50% overestimated and 50% underestimated compared with the results from mass balance, probably due to nonideal breakthrough behaviors of microbial transport in structured soils. According to Kretzschmar et al. (1997), for columns with ideal breakthrough behaviors of colloid transport and high Peclet numbers, the results estimated from the mass balance method and curve fitting are practically identical, otherwise the mass balance method yields more reliable results. Pang et al. (2005) also demonstrated that spatial removal rates estimated from curve fitting and calculated from spatial concentration data are not significantly different based

Table 2. Removal rates determined from curve-fitting, mass balance and peak concentration methods for selected breakthrough curves of fecal coliforms and bacteriophages (data from Pang et al., 2008)

Experiment	Length	Peak	CXTFIT curve-fitting results				HYDRUS	Removal rate λ (log_{10}/m)		
Soil-microbe‡	*x*	concentration	*V*	*D*	*k*	r^2	k_{mim}§	CXTFIT	Mass balance	Peak concentration
	m	C_p/C_o	cm/h	cm^2/h	h^{-1}		h^{-1}	[Eq. 3]	[Eq. 6]	[Eq. 5]
Waikiwi -phg1	0.47	7.59E-02	11.68	17.10	0.49	0.81	0.38	1.83	2.54	2.38
Waikiwi -phg2	0.47	7.80E-02	10.62	13.95	0.44	0.86	0.38	1.79	2.40	2.36
Waikiwi -phg3	0.47	1.07E-01	10.57	11.52	0.41	0.82	0.36	1.69	2.39	2.07
Waikiwi -FC1	0.47	7.87E-02	11.27	16.18	0.51	0.73	0.41	1.96	2.71	2.35
Waikiwi -FC2	0.47	8.54E-02	10.83	15.29	0.51	0.59	0.42	2.05	2.94	2.27
Waikiwi -FC3	0.47	4.88E-02	10.53	4.03	0.48	0.92	0.43	1.97	2.80	2.79
Waikoikoi-phg1	0.50	7.68E-02	8.51	9.43	0.33	0.96	0.34	1.71	2.19	2.23
Waikoikoi-phg2	0.50	4.51E-02	9.20	5.85	0.42	0.99	0.42	1.99	2.66	2.69
Waikoikoi-phg3	0.50	9.02E-02	8.48	15.20	0.30	0.95	0.31	1.55	1.97	2.09
Templeton-phg1	0.40	9.50E-02	5.67	3.87	0.36	0.92	0.32	2.75	2.52	2.56
Templeton-phg2	0.40	1.81E-01	6.99	4.78	0.34	0.90	0.35	2.10	1.98	1.85
Templeton-phg3	0.40	2.39E-01	7.22	11.06	0.30	0.95	0.32	1.78	1.54	1.56
Waitarere-phg1	0.70	2.06E-02	2.39	4.45	0.12	0.92	0.13	2.11	1.92	2.41
Waitarere-phg2	0.70	3.51E-02	2.07	4.12	0.08	0.92	0.07	1.75	1.58	2.08
Waitarere-phg3	0.70	9.49E-03	2.16	4.65	0.15	0.71	0.15	3.05	2.57	2.89
Waitarere-FC1	0.70	2.50E-02	2.41	4.47	0.11	0.92	0.10	1.97	1.80	2.29
Waitarere-FC2	0.70	4.27E-02	2.09	4.14	0.08	0.92	0.07	1.61	1.46	1.96
Waitarere-FC3	0.70	1.15E-02	2.18	4.68	0.15	0.71	0.15	2.89	2.45	2.77

‡ phg - bacteriophage, FC - fecal coliforms. 1, 2, 3 refers to lysimeter number.

§ k_{mim} is the temporal removal rate determined from mobile-immobile water two region model of HYDRUS-1D given in Pang et al (2008).

on an evaluation of 17 sets of field tracer data. They concluded that both the log $(C_p/C_o) \sim x$ graphical approach and the inverse modeling of BTC approach could be used for derivation of removal rates depending on the type of data that are available. The similarity of the results generated by the three different methods suggests that the simple peak-concentration method for continuous inputs could be used to approximate pulse inputs when BTC data are not available, which is often the case for the studies published in the literature. Other researchers (e.g., McKay et al., 2000) have also used the simple peak-concentration method for removal rate estimations.

With the exception of soil lysimeters from which leachates are fully captured, it is very difficult to accurately estimate mass balance from field data. Although the percentage recovery, called recovery efficiency in this paper, is often reported in field studies, it is more likely to relate to concentration rather than to mass.

In the following sections, a λ value is interpreted from the slope of linear fit for a log (C_p/C_o) vs. x plot when there are multiple sampling locations down-gradient of the source. When numbers of sampling locations are limited, a λ value is directly calculated from peak-concentration, or recovery efficiency, or converted from the temporal removal rate (attachment rate + inactivation rate) if modeling results are available. Occasionally, mass balance data are also used. Table 3 gives the notations for the methods and the type of data used in the estimation of removal rates.

Soils

Soils are the unconsolidated minerals and organic materials on the immediate surface of the earth and they serve as a natural medium for the growth of plants. Soil contains humus, earth animals, grass, plant roots, and other marks of biological activity. Well structured soils contain macropores formed by soil fauna (e.g., earthworms, insects, and underground mammals), channels formed by plant roots, cracks and fissures formed during the shrinkage of clay soils and freeze/thaw cycles, and natural soils pipes caused by erosion (Jamieson et al., 2002). Soils are normally unsaturated and the air–water interface interaction plays a very important role in enhanced microbial removal. In this study, the lower boundary of soil is arbitrarily set at 1 m, which is approximately the root-zone depth for most plants.

With overwhelming evidence of the important effects of soil structure (especially macropores) on microbial transport (Jamieson et al., 2002; McMurry et al., 1998), greater numbers of more recent studies have used large (>30 cm in length) undisturbed soil lysimeters or in situ intact soil blocks to investigate microbial transport in structured soils (Aislabie et al., 2001; Carlander et al., 2000; Jiang et al., 2008; Karathanasis et al., 2006; McLeod et al., 2001, 2003, 2004; Pang et al., 2008; Roodsari et al., 2005; Shelton et al., 2003).

Some Field and Lysimeter Studies on Microbial Transport in Soils

A wide range of key New Zealand soils from 12 field sites has been investigated by Aislabie et al. (2001), McLeod et al. (2004, 2003, 2001) and Pang et al. (2008) for their ability to attenuate fecal coliforms and bacteriophages. Leaching experiments using intact vegetated soil cores were performed (in triplicate) with a pulse of dairy shed effluent spiked with *Salmonella* bacteriophages and Br. These studies demonstrate that microbial transport in highly structured soils is predominantly controlled by macropores, and that even a very small amount of water can lead to a rapid and significant microbial leaching through bypass flow, particularly in clayey soil and clayey silt loam. Most

of these experimental data were evaluated by Pang et al. (2008) using the HYDRUS-1D mobile-immobile two-region model. The modeling results suggest that, in comparison with the Br solute tracer, microbial transport in most soils showed velocity enhancement, less dispersion, and a much smaller mobile water content (on average, only 19% of the total water content), and that soil structure (macroporosity) plays the most important role in microbial transport, while soil lithology has the greatest influence on microbial attenuation. It is also confirmed that the general pattern of predicted mobile water content agreed with the measured macroporosity, which was positively related to leaching vulnerability but negatively related to dispersivity. Removal rates estimated for these soils based on the peak-concentration method together with relevant information about the soils and experiments are summarized in Table 4.

Jiang et al. (2008) investigated the impact of a number of influencing factors (hydraulic conductivity, irrigation methods, seasonal effects, moisture content, drainage rate, and porosity) on the fecal coliform leaching from dairy shed effluent through a silt loam. Six vegetated soil lysimeters (70 cm in length and 50 cm in diameter) were exposed to field atmospheric conditions. Flood irrigation generally resulted in more bacterial leaching than spray irrigation, and bacterial leaching positively correlated with water moisture content and drainage rate. Greater bacterial leaching was found in the lysimeter with rapid hydraulic conductivity. For the lysimeter with rapid conductivity, there were no obvious effects of the type of irrigation, and bacterial leaching immediately followed effluent application in both flood and spray irrigation, especially in the summer. Bacterial leaching was greater in summer than in autumn because there were more surface cracks present during summer. This was especially true for those lysimeters with higher clay content in the topsoil, where shrinkage cracks can form during summer, promoting preferential flow and facilitating bacterial leaching. Removal rates estimated from the mass balance method for these lysimeters are summarized in Table 5. Table 5 shows that high removal rates relate to lysimeters with low hydraulic conductivities and vice versa. This is because less conductive soils contain fewer macropores and thus would undergo more straining and attachment.

Karathanasis et al. (2006) studied the effectiveness of soils of different textures and depths to remove fecal bacteria eluted from septic effluent. As their study focused on investigating the effect of soil texture, the soil cores collected avoided evident cracks, biochannels, tree roots, rocks, and other inclusions that cause preferential flow. Sod and other organic materials were removed from the soil surface before coring. They found a greater removal of fecal bacteria in fine-textured soils than coarse-textured soils, and that bacterial removal increased with increasing soil depth. The removal rates estimated and relevant information about the soils and experiments are listed in Table 6.

Roodsari et al. (2005) used in situ lysimeters to investigate the effect of vegetation on vertical leaching and surface runoff by bacteria released from surface-applied bovine manure. While bacteria numbers fell in runoff from vegetated plots compared to nonvegetated plots, their vertical transport through clay loam and sandy loam was enhanced in the presence of vegetation. This is because the root channels associated with vegetation would have formed preferential flow paths and thus promoted the vertical transport of bacteria. Table 7 shows that the mass recoveries of fecal coliforms and transport depths under vegetated plots were greater than for nonvegetated plots. However, the effect of vegetation on microbial transport was not observed in the lysimeter study performed by Carlander et al. (2000) with sandy soils. Using *Salmonella* phages to evaluate the

Table 3. Notations for Tables 4 to 16.

Method used for removal rate estimation:	
a	Peak concentration and distance, $\text{Log}(C_p/C_o)/x$
b	The slope of linear fit for the $\log(C_p/C_o)$ $-x$ plot and r^2 measures the goodness of fit
c	Removal rate in natural log
d	Recovery efficiency (or mass recovery) and distance
o	Given attachment rate and pore-water velocity
Data used for removal rate estimation:	
e	Given C_p and distance
f	Given C_p, distance read off diagram
g	Given distance, C_p read off diagram
h	Given distance and recovery efficiency
i	Given removal rate in natural log
j	C_p and distance read off diagram
k	Slope read off from given log$C$–x plot or $\log(C_p/C_o)$ $-x$ plot
l	Slope read from the given linear fit for the log zero moment $-x$ plot
m	C_p provided from the author by personal communication
n	After Pang et al (2005)
Parameter:	
C_p	Peak concentration of microbe (N/L^3)
C_0	Input concentration of microbe (N/L^3)
D	Dispersion coefficient (L^2/T)
d	Mean particle size (L)
d_{10}	The 10th percentile in a cumulative frequency curve of particle size (L)
d_{50}	The 50th percentile in a cumulative frequency curve of particle size (L)
d_{60}	The 60th percentile in a cumulative frequency curve of particle size (L)
I	Hydraulic gradient
K	Hydraulic conductivity (L/T)
K_s	Saturated hydraulic conductivity (L/T)
k	Temporal removal rate or attachment rate (T^{-1})
Q	Flow rate (L^3/T)
R	Retardation factor
r^2 linear	The goodness-of-fit for a linear $\log(C_p/C_o)$ $-x$ function
r^2 log	The goodness-of-fit for a log $\log(C_p/C_o)$ $-x$ function
V	Pore-water velocity (L/T)
x	Transport distance (L)
θ	Porosity
θ_e	Effective porosity
ρ_b	Bulk density (M/L^3)
λ	Spatial removal rate in $\log_{10}$/m, simply expressed as log/m

Abbreviations:			
asl	Above sea level	bgl	Below ground level
DSE	Dairy shed effluent	SW	Sewage
SE	Septic tank effluent	OC	Organic carbon content
CEC	Cation exchange capacity	DOC	Dissolved organic carbon
Con.	Contaminated	Unc.	Uncontaminated
Min.	Minimum	Max.	Maximum
No.	Number of observed dataset	WT	Water table (bgl)
N.R.	Normalized range, calculated from (maximum - minimum)/mean		

Table 4. Microbial removal rates in New Zealand soils derived from large intact vegetated lysimeters irrigated with a pulse of dairy shed effluent.

Reference†	Soil name	Soil texture	Length × diameter	$Q_{Effluent}$	$Q_{Irrigation}$	pH	CEC	Clay	OC	*Ks*	Microbe	λ based on C_p (Log/m)				
												Mean	Min.	Max.	N.R.	No.
			cm	mm/h			cmol/kg	%		mm/h						
1	Atiamuri	Pumice soil	70 × 46	5	5	5.7–6.2	6–21	1–3	0.4–8.1	30–60	*Salmonella* phage	16.61	15.75	17.46	0.10	3
2	Atiamuri	Pumice soil	70 × 46	50	5–10	5.7–6.2	not given	5.00	0.4–8.1	30–60	Fecal coliforms	Complete removal				3
											E. coli	Complete removal				3
											Enterococci	Complete removal				3
1	Waihou	Allophanic soil	70 × 46	5	5	5.0–5.7	12–28	26–34	1.7–6.8	110–120	*Salmonella* phage	Complete removal				3
2	Waihou	Allophanic soil	70 × 46	50	5–10	5.8–6.0	not given	20–25	1.5–6.5	200–1200	Fecal coliforms	5.48	5.22	5.75	0.10	2
											E. coli	5.34	5.04	5.63	0.11	2
											Enterococci	5.16	5.05	5.28	0.04	2
3	Manawatu	Fine sandy loam	70 × 46	5	5	5.5–6.4	8.7–11.3	10–20	0.5–2.0	17–43	*Salmonella* phage	2.98	2.40	3.28	0.29	3
											Fecal coliforms	9.34	8.88	9.56	0.07	3
1	Waitarere	Recent sandy soil	70 × 46	5	5	5.0–5.7	3–19	2–6	0.6–6	100–200	*Salmonella* phage	2.46	2.08	2.89	0.33	3
											Fecal coliforms	2.34	1.96	2.77	0.35	3
4	Waikiwi	Silt loam	46 × 46	5	5	5.7–6.1	9.1–15.4	12–24	1.6–4.1	4–50	*Salmonella* phage	2.27	2.07	2.38	0.14	3
											Fecal coliforms	2.47	2.27	2.79	0.21	3
4	Waikoikoi	Silt loam	50 × 46	5	5	5.9–6.0	9–14.2	1.2–3.2	16–23	4–15	*Salmonella* phage	2.34	2.09	2.69	0.26	3
5	Templeton	Deep Silt loam	40 × 46	5	5	5.6–5.9	6.7–15.2	15–24	0.3–3.1	14.5–385	*Salmonella* phage	1.99	1.56	2.56	0.50	3
											Fecal coliforms	4.09	1.28	1.80	0.13	2
6	Templeton	Deep Silt loam	70 × 50	25–40	variable	4.1–5.8	not given	5.9–13.0	6.1–7.2	33–250	Fecal coliforms	3.66	0.12	6.25	1.67	8
				ponding	variable	4.1–5.8				33–123	Fecal coliforms	4.29	3.34	5.94	0.61	4
5	Lismore	Shallow silt loam over gravels	70 × 46	5	5	5.6–5.9	8.3–11.7	12–24	0.8–2.2	114–723	*Salmonella* phage	1.98	0.99	2.53	0.78	3
											Fecal coliforms	4.04	2.42	6.49	1.01	3
3	Rangiro	Silty clay loam				4.80	20.4–25.9	70–80	1.2–8.0	5–88	*Salmonella* phage	2.80	1.87	4.18	0.83	3
											Fecal coliforms	3.61	2.77	5.16	0.66	3
3	Hamilton	Clay loam	70 × 46	5	5	4.9–5.3	9.9–17.2	30–79	0.8–3.0	13–200	*Salmonella* phage	1.80	1.59	2.15	0.31	3
											Fecal coliforms	2.64	2.08	3.17	0.41	3
2	Te Kowhai	Clayey silt loam	75 × 46	50	5–10	5.7–6.6	not given	30–40	0.4–6.9	200–300	Fecal coliforms	0.55	0.40	0.69	0.53	2
											E. coli	0.54	0.42	0.65	0.44	2
											Enterococci	0.27	0.20	0.33	0.49	2
1	Netherton	Clayey soil	70 × 46	5	5	5.1–6.1	27–32	52–69	0.8–5.4	30–50	*Salmonella* phage	0.97	0.12	2.08	2.03	3
2	Netherton	Clayey soil	70 × 46	50	5–10	4.8–5.5	not given	65–70	0.8–5.5	30–50	Fecal coliforms	0.41	0.00	0.83	2.00	2
											E. coli	0.34	0.00	0.69	2.00	2
											Enterococci	0.79	0.72	0.86	0.18	2

† 1, McLeod et al. (2001); 2, Aislabie et al. (2001); 3, McLeod et al. (2004); 4, McLeod et al. (2003); 5, Pang et al. (2008). 1-5 all indoor lysimeters, in triplicates. 6, Jiang et al. (2008), outdoor lysimeters, exposed to field climatic conditions (rain + irrigation). Effluent spray irrigation 25 to 40 mm/h and ponding for flood irrigation.

Table 5. The effect of saturated hydraulic conductivity, irrigation method and season on leaching of fecal coliforms from dairy shed effluent through intact vegetated soil lysimeters (based on data of Jing et al., 2008).

							Summer			Autumn		
Core†	Texture	pH	OC	Clay	θ	*Ks*	Method	Mass recover	λ‡	Method	Mass recover	λ‡
			——%——			mm/h		%	Log/m		%	Log/m
A	Silt loam	4.40	6.70	13.00	0.51	42	Spray	0.10	4.27	Spray	0.00	7.14
B	Sandy loam	5.00	6.20	5.90	0.49	123	Flood	0.48	3.31	Spray	2.70	2.24
C	Loam	5.80	7.20	10.90	0.48	250	Spray	49.80	0.43	Spray	18.00	1.06
D	Silt loam	4.70	7.20	10.80	0.48	41	Spray	0.03	5.05	Flood	0.01	5.94
E	Silt loam	4.70	6.30	7.60	0.52	110	Flood	0.54	3.24	Spray	0.00	7.14
F	Silt loam	4.10	6.10	8.30	0.50	33	Spray	0.01	6.10	Flood	0.61	3.16

† The lysimeters were exposed to field climatic conditions. Effluent spray irrigation rates: 25-40 mm/h and ponding for flood irrigation, respectively.

‡ Removal rates are estimated from mass balance method.

Table 6. Removal rates of fecal bacteria in soils from Kentucky derived from intact sod-removed lysimeters† irrigated with septic tank effluent estimated from data of Karathanasis et al. (2006).

Soil name	Texture	pH	OC	Clay	CEC	*Q*	*Q*	Base saturation	$\lambda_{Fecal\ Coliforms}$‡	r^2 linear	r^2 log	$\lambda_{Fecal\ streptococci}$‡	r^2 linear	r^2 log
			——%——		cmol/kg	mL/h	mm/h		log/m			log/m		
Yeager 1	loamy sand	4.9	0.6	7.0	2.3	45	0.09	0.11	6.66	0.99	0.74	6.07	0.99	0.84
Yeager 2	loamy sand	4.8	0.6	10.0	2.5	45	0.09	0.10	1.38	0.44	0.79	1.37	0.47	0.47
Bruno	sandy loam	6.2	3.4	12.0	6.2	35	0.07	0.50	3.89	0.67	0.93	5.02	0.85	0.47
Lily	loam	4.7	0.4	16.0	5.4	35	0.07	0.07	4.89	0.94	0.92	5.50	0.99	0.89
Pope	sandy loam	4.8	0.8	18.0	4.7	35	0.07	0.14	2.63	0.64	0.59	2.24	0.62	0.63
Ashton	sandy loam	6.2	2.0	20.0	18.0	20	0.04	0.18	3.15	1.00	0.75	3.03	0.94	0.67
Nolin	sandy loam	5.7	2.5	20.0	14.9	20	0.04	0.33	5.13	0.77	1.00	5.17	0.75	1.00
Shelocta	clay loam	5.2	0.7	28.0	8.5	20	0.04	0.19	0.81	0.15	0.56	1.75	0.22	0.63
Lowell	silty clay/clay	5.7	0.3	52.0	19.2	15	0.03	0.45	2.44	0.21	0.58	2.76	0.34	0.71
Maury	clay	6.0	0.1	42.0	16.5	15	0.03	0.22	3.67	0.95	0.91	6.04	0.97	0.81

† The intact lysimeters are 30, 45, 60 cm in length and 25 cm in diameter. Lysimeters were presaturated prior to effluent irrigation and experiments were carried out under anaerobic condition.

‡ Removal rates were determined from the slope of linear $\log(C_p/C_o)$ vs. depth plots (total four data points including the origin).

impact of viruses on groundwater quality from wastewater irrigation, Carlander et al. (2000) found there were no clear differences between the willow-cropped and the nonwillow cropped lysimeters, and that willow plants did not seem to facilitate leaching of phages in the sandy soil by creating root channels. The contrasting findings of the above two studies suggest that the level of significance of root presence on microbial leaching is affected by both soil's structure stability and textural properties. Using the given mass recoveries, removal rates for these studies are calculated from the mass balance method and listed in Table 7. Table 7 also lists the removal rates estimated from a few other studies using large intact lysimeters.

Sandy soils are often chosen in infiltration basins as an effective filtering media for wastewater treatment. This is shown in the removal rates for soil treatment systems (Table 8) estimated from the field studies of Nicosia et al. (2001), Schaub and Sorber (1977), and Van Cuyk et al. (2004). The removal rates estimated from the field data of Nicosia et al. (2001) suggest that for the same soil media and microbial tracer, the removal rate reduces as the hydraulic load increases. This is a similar finding to that of Smith et al. (1985) and Warnemuende and Kanwar (2002) who observed that bacterial transport increased with faster application rates.

Nicosia et al. (2001) and Shadford et al. (1997) noted that rainfall significantly increased leaching of phages and bacteria under infiltration bed systems. During high rainfall events, the microbial retaining efficiency of the filter beds reduces as the high level of infiltration caused by the rain increases the soil water content which in turn facilitates microbial leaching. In addition, rainfall lowers the ionic strength of pore fluid and thus promotes microbial transport though soils (Bitton and Harvey, 1992). This is expected as lowering ionic strength would increase the thickness of electrostatic double layer, enhancing the magnitude of electrostatic repulsion between the microbial particles and mineral surfaces, thus prohibiting deposition of microbial particles and promoting microbial leaching. This effect of ionic strength on microbial attachment has been reported in many studies (Bolster et al., 2006; Choi et al., 2007; Fontes et al., 1991; Mills et al., 1994; Scholl et al., 1990).

Table 9 lists removal rates estimated for marshland and hillslope soils. The removal rates estimated from the field data of Watson and Rusch (2002) and Rahe et al. (1978) also shows a decline in removal rate with hydraulic loading. However, this trend is not very clear in terms of the removal rates derived from the study of Richardson and Rusch (2005). Note that the results of Table 9 include a significant influence from horizontal transport.

Summary and Recommendations for Soils

Categorizing by soil types, the information detailed in Tables 4–9 is abstracted in Table 10. The following comments can be made to summarize the microbial removal rates for soils calculated in this study (Tables 4–10):

1. Microbial removal rates are generally in the order of 10° log/m (i.e., a few log/m) for most soil types, 10^{1} log/m

Table 7. Removal rates of microbes in unsaturated soils derived from large intact lysimeters with manure, sludge, and tracer solution applied.

Reference†	Soil	Length × diameter	Contaminant source	$Q_{Irrigation}$	pH	OC	Microbe	Total mass recovery	Depth	Removal rate λ (Log/m)				No.
										Mean	Min.	Max.	N.R.	
		cm		mm/h		%		%	cm					
1	Vegetated clay loam	300 deep	Bovine manure	61	5.3–6.5	1.8–3.6	Fecal coliform	90	10	0.46				1
1	Bare sandy loam	300 deep	Bovine manure	61	5.9–6.5	0.8–2.6	Fecal coliform	33	20	2.41				1
1	Vegetated sandy loam	300 deep	Bovine manure	61	5.9–6.5	0.8–2.6	Fecal coliform	11	60	1.60				1
2	Stony silt loam	90×60	Bovine manure	71	5.0	0.3–2.0	Fecal coliform		90	2.48	1.61	2.69	0.43	6
3	Fine sandy loam	54×5	Sewage sludge	7.64	5.1–5.6		Vaccine poliovirus	0.1–0.2	54	5.26	4.97	5.54	0.11	2
4	Loamy sand	100×30	Trace solution	1.83		1–2	*Salmonella* phage	0.001–0.183	100	3.76	2.74	4.87	0.57	6

† 1, Roodsari et al. (2005): in-situ lysimeters. Mass recovery and depths are given; 2, Shelton et al. (2003): soil contained stones of shale, sandstone, and siltstone. C_p/C_o was read from Fig. 3 of original paper, K=1.5–15.2 cm/h. Average removal rate is calculated from given V = 0.5 cm/h and k = 0.0054/h, λ = 0.0054/0.5 × 100 × 2.3 = 2.48 log/m; 3, Bitton et al. (1978): 2.5 cm of digested sewage sludge seeded with Poliovirus type 1 was applied to the core (θ = 0.43–0.50) followed by rainwater elution with a 2.5-cm head on the column. Mass recover was calculated from original data given in the paper. $Q_{Irrigation}$ is calculated from flux 2.5 mL/min and surface area of the core; 4, Carlander et al. (2000): topsoil 25 cm sandy loam with clay 8%, subsoil 65-cm structureless loamy sandy with clay 1%. Water saturation level was above field capacity. Lysimeters were irrigated with plant nutrient solution. Mass recovery was given in the paper.

Table 8. Removal rates of microbes in soil treatment systems.

Reference	Soil media and property	Contaminant source	Thickness of soil	WT bgl	*Ks*	Hydraulic loading	Microbe	Removal rate λ (log/m)			r^2 linear	r^2 log	No.	Method & data
								Mean	Min.	Max.				
			——m——		m/d	cm/d								
Nicosia et al. (2001)	Fine-very fine sand ρ_b = 1.5 g/cm³, pH = 6.2–7	Septic tank effluent infiltration cell, Florida.	0.60	0.60	5.67–6.69	6.30	PRD1	7.02	5.02	9.01			2	a,e
				below cells		3.20	PRD1	11.37	9.06	13.68			2	a,e
Schaub and Sorber (1977)	Silty sands and gravel Fort Devens, MA	Rapid sewage infiltration basins operated for more than 30 yr. Removed 1.22-m soil and backfilled	0.76	>1.22	8.64		f2 bacteriophage	2.19	1.31	2.86	0.41–0.49	0.07–0.92	5	b, e
			0.76	>1.22			Fecal coliform	8.28			0.90	0.97	3	b, e
			0.76	>1.22			Fecal streptococci	4.81	2.31	8.58	0.88–0.99	0.53–0.96	5	b, e
Van Cuyk et al. (2004)	Medium sand, d_{10} = 0.22 mm, d_{60} = 0.6 mm, pH = 6.8, OC = 0.017%, Colorado	Septic tank effluent infiltration cells. Intact soil cores from five sites, mature systems	0.15	0.90	27.6	0.5–2.7	MS2 or PRD1	15.8	11.63	20.00			5	d,h
			0.30	0.90			MS2	8.34	7.67	9.00			5	d,h
				below infiltration surface			PRD1	8.41	6.82	10			5	d,h

Table 9. Removal rates of bacteria transport in saturated soils estimated from field experiments carried out in wetlands and hillslopes.

Reference	Aquifer	Soil properties	Source	$Q_{Injection}$	x	Microbes	λ	r^2 linear	r^2 log	No.	Method & data
				L/min	m		log/m				
Watson and Rusch (2002)	Marshland upwelling system	Port Fourchon, Louisiana, tidal Spartina salt marsh, Scatlake soils, very fine, very poorly drained, semifluid, clayey, mineral soils, WT = +0.3–0.15 m	Effluent	0.9	4.8	*E. coli*	1.28	0.87		8	b,i
				1.9	4.8	*E. coli*	1.11	0.70		10	b,i
				3.8	4.8	*E. coli*	0.99	0.72		11	b,i
Richardson and Rusch (2005)	Marshland upwelling system	Saline Juncus marsh, Moss Point, Mississippi, Scatlake soils, poorly drained clayey, semifluid, mineral soils, high watertable, d_{50} = 0.04–0.16 mm, d_{60}/d_{10} = 1.08, groundwater pH = 6.5–6.8, temperature 22–23 °C, water-table fluctuated (could be 0.3 m), have a high shrink- swell potential due to the clay component	Effluent	1.9	3.0	Fecal coliforms	2.85	linear		6	b,i
				2.8 (30 min/3 h)	6.0	Fecal coliforms	2.53	Linear		9	b,i
				2.8 (15 min/h)	7.0	Fecal coliforms	2.42	linear		10	b,i
				1.9	D =1.5	Fecal coliforms	3.29			6	a,e
				2.8 (30 min/3 h)	D =1.5	Fecal coliforms	2.64			10	a,e
				2.8 (15 min/h)	D =1.5	Fecal coliforms	2.54			4	a,e
				5.5	D = 1.5	Fecal coliforms	3.88			2	a,e
				1.9	D = 2.3	Fecal coliforms	2.24			6	a,e
				2.8 (30 min/3 h)	D = 2.3	Fecal coliforms	1.74			10	a,e
				2.8 (15 min/h)	D = 2.3	Fecal coliforms	1.65			4	a,e
				5.5	D = 2.3	Fecal coliforms	2.53			2	a,e
				1.9	D = 3.0	Fecal coliforms	1.42			6	a,e
				2.8 (30 min/3 h)	D = 3.0	Fecal coliforms	1.24			10	a,e
				2.8 (15 min/h)	D = 3.0	Fecal coliforms	1.19			4	a,e
				5.5	D = 3.0	Fecal coliforms	1.81			2	a,e
Rahe et al. (1978)	Silty clay and clay	Hazelair Hillslope, WT = 0.153 m, I = 0.1, K = 0.04–17.97 cm/h, pH = 5.6–5.7	Tracer	0.15	15.0	*E. coli*	0.32	0.76	0.20	5	b,g
				0.15	20.0	*E. coli*	0.36	1.00	0.94	3	b,e

Table 10. Summary of microbial removal rates for different soils.

									λ based on C_p		
Soil texture	**Source**	***Q***	**pH**	**CEC**	**Clay**	**OC**	***K*$_s$**	**Microbe**	**Mean**	**Min.**	**Max.**
		mm/h		cmol/kg	———%———		mm/h		———Log/m———		
Pumice soil	DSE	5	5.7–6.2	6–21	1–3	0.4–8.1	30–60	*Salmonella* phage	16.61	15.75	17.46
								Complete removal in fecal bacteria			
Allophanic soil	DSE	5–10	5.8–6.0		20–25	1.5–6.5	200–1200	Fecal coliforms	5.48	5.22	5.75
								E. coli	5.34	5.04	5.63
								Enterococci	5.16	5.05	5.28
								Complete removal in *Salmonella* phage			
Fine sandy loam	DSE	5	5.5–6.4	8.7–11.3	10–20	0.5–2.0	17–43	*Salmonella* phage	2.98	2.40	3.28
								Fecal coliforms	9.34	8.88	9.56
	SW sludge	7.64	5.1–5.6					Poliovirus	5.26	4.97	5.54
Fine-very fine sand	SW	1.33–2.63	6.2–7				5.67–6.69	PRD1	9.19	5.02	13.68
Medium sand	SE	0.21–1.13	0.07			0.017	27.6	MS2 or PRD1	10.85	6.82	20.00
Recent sandy soil	DSE	5	5.0–5.7	3–19	2–6	0.6–6	100–200	*Salmonella* phage	2.46	2.08	2.89
								Fecal coliforms	2.34	1.96	2.77
Loamy sand	SE	0.09	4.8–4.9	2.3–2.5	7–10	0.6		Fecal coliforms	4.02	1.38	6.66
		0.09	4.8–4.9	2.3–2.5	7–10	0.6		Fecal streptococci	3.72	1.37	6.07
	Trace	1.83				1-2		*Salmonella* phage	3.76	2.74	4.87
Sandy loam	SE	0.04–0.07	4.8–6.2	4.7–6.2	12–18	0.8–3.4		Fecal coliforms	3.70	2.63	5.13
	DSE	25–flood	5.00		5.90	6.20	123	Fecal coliforms	2.78	2.24	3.31
	SE	0.04–0.07	4.8–6.2	4.7–6.2	12–18	0.8–3.4		Fecal streptococci	3.87	2.24	5.17
Bare sandy loam	Cow manure	61	5.9–6.5			0.8–2.6		Fecal coliforms	2.41		
Vegetated sandy loam	Cow manure	61	5.9–6.5			0.8–2.6		Fecal coliform	1.60		
Silty sands and gravel	SW						8.64	f2 bacteriophage	2.19	1.31	2.86
								Fecal coliform	8.28		
								Fecal streptococci	4.81	2.31	8.58
Silt loam	DSE	5	5.7–6.1	9.0–15.4	1.2–24	1.6–23	4–50	*Salmonella* phage	2.30	2.07	2.69
	DSE	5	5.7–6.1	9.1–15.4	12–24	1.6–4.1	4–50	Fecal coliforms	2.47	2.27	2.79
	DSE	25–40	4.1–4.7		7.6–13	6.1–7.2	33–110	Fecal coliforms	6.00	4.27	7.14
	DSE	flood	4.1–4.7		7.6–10.8	6.1–7.2	33–110	Fecal coliforms	4.11		
Deep silt loam	DSE	5	5.6–5.9	6.7–15.2	15–24	0.3–3.1	14.5–385	*Salmonella* phage	1.99	1.56	2.56
	DSE	variable	4.1–5.8		5.9–13.0	6.1–7.2	33–250	Fecal coliforms	4.00	0.12	6.25
Shallow silt loam over gravels	DSE	5	5.6–5.9	8.3–11.7	12–24	0.8–2.2	114–723	*Salmonella* phage	1.98	0.99	2.53
								Fecal coliforms	4.04	2.42	6.49
Stony silt loam	Cow manure	71	5.0			0.3–2.0		Fecal coliform	2.48	1.61	2.69
Silty clay loam	DSE		4.80	20.4–25.9	70–80	1.2–8.0	5–88	*Salmonella* phage	2.80	1.87	4.18
								Fecal coliforms	3.61	2.77	5.16
Silty clay/clay	SE	0.03	5.7	19.2	52.0	0.3		Fecal coliforms	2.44		
		0.03	5.7	19.2	52.0	0.3		Fecal streptococci	2.76		
	Tracer		5.6–5.7				0.04–18.0	*E. coli*	0.34	0.32	0.36
Clay	SE	0.03	6.0	16.5	42.0	0.1		Fecal coliforms	3.67		
		0.03	6.0	16.5	42.0	0.1		Fecal streptococci	6.04		
Clay loam	DSE	5	4.9–5.3	9.9–17.2	30–79	0.8–3.0	13–200	*Salmonella* phage	1.80	1.59	2.15
								Fecal coliforms	2.64	2.08	3.17
	SE	0.04	5.2	8.5	28.0	0.7		Fecal coliforms	0.81		
		0.04	5.2	8.5	28.0	0.7		Fecal streptococci	1.75		
	Cow manure	61	5.3–6.5			1.8–3.6		Fecal coliform	0.46		
Clayey silt loam	DSE	5–10	5.7–6.6		30–40	0.4–6.9	200–300	Fecal coliforms	0.55	0.40	0.69
								E. coli	0.54	0.42	0.65
								Enterococci	0.27	0.20	0.33
Clayey soil	DSE	5	5.1–6.1	27–32	52–69	0.8–5.4	30–50	*Salmonella* phage	0.97	0.12	2.08
		5–10	4.8–5.5	not given	65–70	0.8–5.5	30–50	Fecal coliforms	0.41	0.00	0.83
								E. coli	0.34	0.00	0.69
								Enterococci	0.79	0.72	0.86
Loam	SE	0.07	4.7	5.4	16.0	0.4		Fecal coliforms	4.89		
		0.07	4.7	5.4	16.0	0.4		Fecal streptococci	5.50		
	DSE	25–40	5.80		10.90	7.20	250	Fecal coliforms	0.75	0.43	1.06
Marshland	SW							*E. coli*	1.13	0.99	1.28
	SW		6.5–6.8					Fecal coliforms	2.38	1.19	3.88

or greater for allophanic and pumice sand soils, but could be down to 10^{-1} log/m for clayey soil, clay loam, and clayey silt loam.

2. Of all soil types investigated in this study, allophanic and pumice sand soils have the greatest capacity to remove both bacteria and phages. This is because allophanic clays have a net positive charge when soil pH is below 6.0, which is their isoelectric point (Cooper and Morgan, 1979). The pH values for allophanic and pumice topsoils in the field are typically < pH 6 (Table 4), therefore they have an affinity for net negatively charged bacteria and phages. In addition, allophane has a very large surface area, 700 to 900 m^2 g^{-1} (Aislabie et al., 2001), further enhancing microbial removal with the volcanic soil media.
3. Volcanic soils are followed by fine sandy loam, sandy loam, and loamy sand for efficiency in microbial removal. Fine sandy loam is very effective at removing bacteria probably due to straining, but it is relatively ineffective at removing phages (Table 4).
4. Silt loam, shallow and deep silt loams have moderate capacities in microbial removal.
5. The worst soils for microbial removal are clayey soils and clay loam. Although clay particles are very effective at filtering microbial particles under conditions of ideal matrix flow (Keswick and Gerba, 1980), clay soils under field conditions are susceptible to shrinking and cracking forming macropores and preferential flow paths (Carlander et al., 2000). Rapid microbial leaching immediately after effluent irrigation is often observed in structured clayey soil, clayey silt loam, and clay loam. Similarly, Carlander et al. (2000) also noted that phage transport was generally more rapid and had a much lower retention in clay soils than in sand soils in their field lysimeter study. This suggests that under field conditions, the effect of soil structure (i.e., macropores) often overrides the effect of texture on microbial removal. A clay soil core with many cracks and channels might favor microbial transport compared with a sandy soil core with a more homogenous pore structure (Guimaraes et al., 1997). With intact soil cores, there is sometimes no relationship between soil texture and microbial transport (Guimaraes et al., 1997; Smith et al., 1985).
6. Removal rates are more variable (refer to normalized range, NR values in Tables 4 and 7) in soils containing clay and gravels (clayey soil, silty clay loam, clay loam, silt loam-over-gravels, and deep silt loam) than fine textured and volcanic soils (silt loam, fine sand loam, recent sandy soil, allophanic soil, and pumice sand soils).
7. For a specific soil, the removal rate for fecal coliforms is generally greater than that for bacteriophages, but they are within the same order of magnitude. Removal rates for fecal coliforms, *Escherichia coli*, *streptococci,* and *enterococci* are similar.
8. For a particular soil, removal rates determined from experiments with flood irrigation are lower due to a greater transport but less variable than those determined from spray irrigation. This is because soil drainage is greatly in excess of soil moisture for flood irrigation; whereas for spray irrigation the amount applied may or may not exceed the soil moisture deficit, depending on the time of year, irrigation method, irrigation rate, and uniformity of application.
9. For a particular soil (e.g., Templeton Soil in Table 4), removal rates determined from indoor lysimeters are less variable than those determined from outdoor lysimeters although they are still within the same order of magnitude. Soil structure can change with seasons. This is particularly relevant to the soils with higher clay content in the topsoil as shrinkage cracks can form during summer but can close up during wet seasons.

The information on microbial removal rates in soils can be used to select the desirable soil media for effluent disposal, for example, the selection of backfill materials in septic tank disposal trenches, soil treatment systems, effluent infiltration basins, and field sites for effluent irrigation. The most desirable soil media for effluent disposal are volcanic soils, and followed by sandy soils, which is implied by the derived removal rates. In contrast, clayey soils and gravely soils are not desirable for effluent disposal.

It is a common practice to recycle nutrients to fertilize pasture, forests, and crops through applying effluent and animal manures onto land. With appropriate management practices, the efficiency of soils for microbial removal could improve. When soil macropores are disturbed, bacterial transport is substantially decreased (Abu-Ashour et al., 1998). McMurry et al. (1998) found that the volume of water and the duration of irrigation required to elute the maximum concentration of fecal coliforms was significantly greater in tilled soil blocks than in sod-covered soil blocks. Tillage reduces microbial transport by disturbing preferential flow paths (Jamieson et al., 2002; McMurry et al., 1998), and could be used to slow microbial leaching through the soil profile (McMurry et al., 1998). Reducing the rate of irrigation is another good way to minimize microbial leaching.

As the microbial removal rates for soils previously described are derived from studies with soil depths of <1 m, these removal rates should not be extrapolated to depths >1 m; beyond 1 m, removal rates for vadose zone media should be considered.

Vadose Zones

Vadose zones are defined in this paper as unsaturated subsurface media below soils, and are either comprised of earthy materials or hard rock devoid of animals, roots, or any other markers of biological activity. Biological activity is lower in vadose zones compared with soils. As they are covered by soils, the effect of the air–water interface interaction is expected to be not as great as in soils, and thus removal rates should be lower in vadose zone media than it is in soils. The lower boundaries of vadose zones are generally groundwater tables, and their upper boundaries are arbitrarily set at 1 m in this paper.

Some Field Studies on Microbial Transport in Vadose Zones

Fewer studies have been undertaken on microbial removal in vadose zones compared with microbial removal in soils and groundwater. Many of the studies performed involving vadose zone media relate to wastewater or sewage effluent infiltration basins (Anders and Chrysikopoulos, 2005; Carre and Dufils, 1991; Ho et al., 1992; Jansons et al., 1989; Powelson et al., 1993; Vaughn et al., 1981). Infiltration experiments through vadose zones using tracer solutions (Frazier et al., 2002; Gerba et al., 1991; McKay et al., 1999), septic tank effluent (Pang et al., 2001; Sinton, 1986), and animal effluent (Krapac et al., 2002) are also reported. Most of these studies are limited to the investigation of bacteriophage and fecal coliform removal, while the study by Jansons et al. (1989) provides valuable information on the removal of waste-associated human viruses through vadose zone. Here, microbial removal rates in vadose zones from these studies are estimated, using the methods described previously, and are listed in Table 11, together with their experimental conditions.

The microbial removal rates estimated for vadose zone media are generally in the order of 10^{-1} log/m for clay and silt, sand, sand-gravels, coarse gravels, and fractured chalk and granite (Table 11). Microbial removal rates are in the order of 10° log/m for pumice sand and clay till, and also occasionally for sand. Like the situation for soil media, the best vadose zone media for effluent infiltration are pumice sand and uniform sand.

It is notable that for the same media, the removal rate for viruses and virus indicators (phages) is in the same order of magnitude as that for bacteria, and can either be lower or higher than bacterial removal rates. Viruses are smaller and can survive longer than bacteria in the natural environment and groundwater (Yates et al., 1987) as unlike bacteria, they do not require nutrients in groundwater to survive, thus viruses could have a lower rate of removal by filtration and inactivation. On the other hand, as viruses in sewage effluent are often associated with colloids (Gerba et al., 1978; Hejkal et al., 1981), they could be removed with colloids that are larger than bacteria, and thus result in a greater removal rate than bacteria.

The removal rate for MS2 phage is generally lower than that for PRD-1 phage (Table 11), particularly near the soil surface (Powelson et al., 1993). However at low infiltration rates, the rate of removal for MS2 phage could be slightly greater than that PRD-1 phage because PRD-1 survives for longer than MS2 (Gerba et al., 1991). The removal rates for human viruses in sand media estimated in this paper are in the order of: enteroviruses < echoviruses type 11 < coxsackieviruses B4 < coxsackieviruses B5 = poliovirus type 2 < echoviruses type 24. Jansons et al. (1989) found that waste-associated viruses penetrated much deeper down than seeded vaccine polioviruses. They commented that the populations (thus surface charge) of waste-associated human viruses would be more variable than viruses cultivated in the laboratory, thus viruses with a stronger net negative charge could penetrate deeper to the soil profile.

The removal rates estimated from the studies by Vaughn et al. (1981) and Gerba et al. (1991) appear to increase with decreasing infiltration rates (Table 11), which is consistent with the finding for soils. Powelson et al. (1993) also observed that removal rate of PRD1 increased with lowering infiltration rate. This is because a decrease in the infiltration rate would increase the travel time and reduce volumetric water content, leading to the greater influence of inactivation and the air–water interface on microbial removal. This finding may be useful in managing effluent infiltration. Groundwater contamination from effluent disposal could be minimized by controlling the infiltration rate, for example, by periodically drying and wetting the infiltration basins or disposal trenches.

Most wastewater infiltration basins listed in Table 11 had operated for many years and the vadose zone media were organically contaminated. Under such conditions, waste-associated viruses and bacteria are often associated with organic materials. Viruses and bacteria may compete with organic materials for attachment to solid surfaces (Harvey et al., 1989; Johnson and Logan, 1996). The adsorption of viruses and bacteria onto organic materials in the effluent may protect them from inactivation (Tate, 1978). This negative influence of natural organic materials on microbial removal was encompassed in the monitoring data used to calculate removal rates.

As infiltration basins are often under conditions of surface-ponding, microbial transport is often under forced hydraulic gradients and the vadose zone media might be close to saturation. However, unlike transport processes in groundwater which occurs largely in a horizontal direction within an aquifer layer, transport processes in vadose zones occur vertically mostly, and perpendicularly to lithological units. Furthermore, geochemical and physical conditions of vadose zones are generally very different from those of unsaturated zones even for the same lithologic units. Therefore, it is expected that the removal rates derived from groundwater do not apply to similar vadose zone media, even when they are close to saturation. The removal rates for groundwater systems will be examined in the next section.

Aquifers

Many field studies investigating microbial transport in groundwater are reported in the literature. Pang et al. (2005) have already compiled microbial removal rates in some sand, sand-gravels, and gravel aquifers. Additional results are presented here to include a wider range of aquifer media (e.g., karst limestone aquifers and fractured rocks) and aquifer conditions (e.g., bank filtration). The removal rates estimated for a range of aquifer media under various aquifer conditions are summarized in Tables 12–15, and the major findings are described below.

Patterns of Removal Rates in Different Aquifers

The removal rates estimated for aquifers are much more variable compared with those for soils and vadose zone media, and they depend on the type of aquifer and its hydraulic characteristics, the transport scale, and duration of contamination. The following patterns were identified (note that the research cited includes different types of organisms):

Sand Aquifers (Flow Velocity <2 m/d)

- 10° log/m for pumice sand aquifers from the studies of Pang et al. (1996) and Wall et al. (2008) performed in New Zealand,

Table 11. Removal rates of microbes in vadose zone media.

Reference	Vadose zone media and property	Contaminant source	Thickness of vadose zone	WT bgl	$Q_{Infiltration}$	Microbe	Removal rate λ			r^2	r^2	No. of obs	Method and data
							Mean	Min.	Max.	Linear	log		
			m		m/d or m^3/d†		log/m						
Krapac et al. (2002)	Silty clay loam	Leaking of a deep pit of pig manure.	5.00	5 m below pit		Fecal streptococcus	0.88					2	a, f
Carre and Dufils (1991)	Very fine uniform dune sands	Wastewater infiltration basins	6.00	7.00	147 (80–200)†	Fecal coliforms		0.52††				2	a, e
	d_{10}=0.2 mm, Creances, France		6.00	7.00	147 (80–200)†	Fecal streptococci		0.45††				2	a, e
Ho et al. (1992)	Sand (d = 0.18 mm), K_s =16 m/d CEC = 1.5 meq/100g.	Sewage effluent groundwater recharge basins. Flooding and drying.	3.75 (0.75–3.75)		0.1	Fecal coliform	0.84			0.89	0.91	5	b, e
					0.1	Coliphage	0.15			0.96	0.86	5	b, e
Jansons et al. (1989)	Bassenden Sand with high silica content and very low pH Canning Vale, Western Australia	Sewage effluent soakage basins, groundwater recharge site, water depth in basin 1 m	3.00	9.00	4 (0.3–9)	Fecal coliform	0.53			0.77	0.42	6	b, e
			3.00	9.00	4 (0.3–9)	Echoviruses type 11‡	0.37			0.99	0.61	5	b, e
			1.00	9.00	4 (0.3–9)	Echoviruses type 24	1.08			0.97	0.64	3	b, e
			0.50	9.00	4 (0.3–9)	Poliovirus type 2§	0.95			1.00	1.00	2	b, e
			3.00	9.00	4 (0.3–9)	Enteroviruses	0.26			0.66	0.53	5	b, e
			1.00	9.00	4 (0.3–9)	Coxsackieviruses B4	0.48			0.75	1.00	3	b, e
			0.50	9.00	4 (0.3–9)	Coxsackieviruses B5	0.95			1.00	1.00	2	b, e
Anders and Chrysikopoulos (2005)	Fine-coarse sand, California θ = 0.3, d = 0.125–1 mm, ρ_b= 1.86 g/cm^3, V=6 m/d	Wastewater infiltration basin, fully saturated vertical flow. Soil surface was tilled to breakup any hardpan.	1.5 (2.1–3.6)			MS2 phages¶	0.95	0.46	1.43				c, i
			1.5 (2.1–3.6)			PRD-1 phages¶	1.52	0.94	2.09				c, i
Pang et al. (2001)	Pumice sand, ρ_b=1.00 g/cm^3, θ = 0.58.	Sewage effluent injected into a ditch excavated 0.65 m bgl.	1.16 (0.65–1.81)	1.81	0.15	F-RNA phages	4.83					1	a, e
				1.81	0.15	Fecal coliforms	2.66					1	a, e
Vaughn et al. (1981)	Coarse sand, and fine gravel Suffolk, N.Y Contain 1.12% silt and clay	Wastewater infiltration basins	7.62	7.62	18–24	Poliovirus	0.36			0.80	0.71	4	b, e
			7.62	7.62	1.44	Poliovirus	0.23			0.53	0.32	4	b, e
			7.62	7.62	0.24	Poliovirus	0.28			0.75	0.51	4	b, e
			7.62	7.62	0.12	Poliovirus	0.29			0.31	0.16	4	b, e
Powelson et al. (1993)	Coarse sand and gravels with clay lenses. Silt and clay 1.8–12.5%	Sewage effluent infiltration basins	4.3 (0.3–4.6)	4.50	1.80–2.70	MS2 phages#		0.05					d, h
			4.3 (0.3–4.6)	4.50	1.00–10.60	PRD-1 phages#			0.59				d, h
Gerba et al. (1991)	Sandy gravel and coarse sand with clay lenses. Near Santa Cruz River, Arizona. Nearly saturated conditions at high flow rate	Sewage effluent spiked with tracers 2–3 d flooding of a mini-basin of 13.4 m^2	4.58	> 6.1	9.15–15.25	MS2 phages	0.12			0.90	0.76	3	b, g
			4.58	> 6.1	0.92–1.53	MS2 phages	0.53			0.74	0.64	4	b, g
			4.58	> 6.1	9.15–15.25	PRD-1 phages	0.33			0.82	0.86	3	b, g
			4.58	> 6.1	0.92–1.53	PRD-1 phages	0.52			0.46	0.90	4	b, g
Sinton (1986)	Coarse gravels	Septic tank effluent soak holes	9.5 (0.5–10)	12.50		Fecal coliforms	0.44	0.27	0.50			7	a, f
McKay et al. (1999)	Fractured clayey till, Denmark K = 6 m/d, I = 0.014–2.5	Tracer-Infiltration box, 0.15 m constant head. $V_{PRD\text{-}1}$=1.5 m/d, V_{Cl} < 0.05 m/d	3.4 (0.6–4)	6.90	0.03–0.25	PRD-1 phages	1.59			0.99	0.98	3	b, e
Beard & Montgomery (1981)	Fissured chalk, Hampshire, UK	Sewage discharge through soakage and drainage by gravity	14	26-35	drainage	Fecal coliforms	0.32	0.31	0.36			3	d,h
					soakage	Fecal coliforms	0.16	0.14	0.19			3	d,h
Frazier et al. (2002)	Weathered and fractured granite	Tracer solution was applied to soil-rock interface. Soil samples taken from trench	1.05	> 1.05	0.20	MS2 phages	0.89					2	a, e

† For the values marked with †, they have the unit of m^3/d.

‡ Detected in 9 m depth in groundwater 14 m from the infiltration basins.

§ Seeded, not detected beyond 1.5 m depth.

¶ Temporal removal rate determined from modeling given in the paper is 0.48 d^{-1} for PRD-1 and 1.68 d^{-1} for MS2.

Temporal removal rate determined from modeling given in the paper is 0.65 h^{-1} for PRD-1 and 0.23 h^{-1} for MS2.

† † Including 6 m vertical distance in aquifer.

Table 12. Removal rates of microbes in sand aquifers estimated from field experiment data.

Reference	Aquifer	Aquifer properties	Contaminant source	*V*	*x*/depth	Microbe	λ	r^2 linear	r^2 log	No.	Method & data
				m/d	m		log/m				
Schijven et al. (1999)	Dune sand	Castricum, The Netherlands, θ_e=0.35, *d*=0.20–0.24 mm. WT=2.85 m asl, dune reclamation areas, recharge basins	River	1.2–1.7	30	MS2	1.87E-01	0.93	0.99	6	b,g,n
					30	PRD1	1.98E-01	0.97	0.91	6	b,g
Carre and Dufils (1991)	Dune sand	d_{10} = 0.2 mm, WT = 6 m, *K* = 12 m/d	Sewage		150	Fecal coliforms	1.42E-02			2	a,e
					150	Streptococci	5.05E-03			2	a,e
Stewart and Reneau (1982)	Coastal sand	Water table rises to flood disposal trench	Septic tanks		13	Faecal coliforms	1.59E-01	1.00		3	b,e,n
Roser et al. (2005)	Fine sand	Jandakot, Australia, Bassendean fine sand	Septic tanks	0.11–0.14	40	Coliform	5.00E-02	0.64	0.61	5	b,e
					40	*E. coli*	4.77E-02	0.61	0.55	5	b,e
					40	Enterococci	2.46E-02	0.34	0.54	5	b,e
					40	*Clostridium*	4.39E-02	0.72	0.69	5	b,e
					40	*C. perfringens*	2.39E-02	0.54	0.57	5	b,e
Mailloux et al. (2003)	Medium sand	With some gravelly layers and small amount of clay and silts, Oyster, Virginia, *d* = 0.25–0.5 mm, *K* = 14–19 cm/h., θ = 0.34, injected 4.8–5.3 bgl, force gradient		0.5–1.5	14/5.2	*Comamonas DA001*	3.38E-01	0.71	0.31	4	b,j
					14/5.4	*Comamonas DA001*	5.94E-01	0.96	0.43	4	b,j
Zhang et al. (2001)	Medium sand	Oyster, Virginia, *d* = 0.35 (0.18–0.71) mm, force gradient, θ = 0.30 (0.25–0.4), pH = 5.6–5.9, OC = 0.4–0.6 mg/L.	Indigenous bacteria strain	Br: 0.6–1.83 *V*= 0.1	5.5 5.5	*Comamonas DA001* Flagellates	1.94E-01 7.59E-02	0.84 0.84	0.90 0.59	5 4	b,g b,g
van der Wielen et al. (2008)	Coarse sand	The Netherlands, anoxic aquifer, O_2<0.5 mg/L, pH=7.5, θ = 0.32, d_{50}=0.405 mm, beneath confining layer of clay	Tracer	Br: 0.33–0.56	37.7	MS2	1.88E-01	1.00	0.49	4	b,e
			Tracer	Br: 0.33–0.56	37.7	ΦX174	2.18E-01	0.98	0.58	4	b,e
Pang et al. (1996)	Pumice sand	Undisturbed pumice sand core, 1 m long 20 cm diam.	Sewage	0.50	1	Faecal coliforms	3.85E+00			1	d,h
Wall et al. (2008)	Pumice sand	d_{50} = 0.15 mm, WT = 5.3 m, DOC = 0.7 mg/L, pH = 6.2	Tracer	Br: 0.9–1.0	2	MS2	1.85E+00			1	a,e
		Temporal removal rate for *E. coli* 0.24–0.35/h		*E. coli* 0.9–1.5	2	*E. coli*	1.54 (1.46–1.61)			3	a,e

· 10^{-2} to 10^{-1} log/m for sand aquifers from the studies of Carre and Dufils (1991), Mailloux et al. (2003), Roser et al. (2005), Schijven et al. (1999), Stewart and Reneau (1982), van der Wielen et al. (2008), and Zhang et al. (2001).

Sand and Gravel Aquifers (Flow Velocity <3 m/d)

· 10^{-1} log/m for x < 17 m estimated from the studies performed at Cape Cod (Bales et al., 1995; Blanford et al., 2005; Harvey and Garabedian, 1991; Harvey et al., 1993, 1995; Pieper et al., 1997) and those in Frenchtown High School near Missoula (DeBorde et al., 1998a, 1998b), and occasionally 10^{-2} log/m and 10° log/m (Blanford et al., 2005).

· 10^{-3} log/m for x = 210 to 970 m and 10^{-4} log/m for x = 210 to 2930 m from the study of Harvey et al. (1984) performed at Cape Cod, and 10^{-3} log/m for x = 183 m from Schaub and Sorber (1977) performed in Devens, MA.

Sand and Gravel Aquifers in River Bank Filtration (x < 177 m)

· 10^{-2} to 10^{-1} log/m in the River Meuse, the Netherlands (Medema et al., 2000), Ohio River, United States (Wang, 2002; Wang et al., 2000; Weiss et al., 2003, 2005), Wabash River and Missouri River (Weiss et al., 2005).

Gravel Aquifers (Fast Flow > 11 m/d)

· 10^{-2} to 10^{-1} log/m for uncontaminated aquifer from DeBorde et al. (1999) and Woessner et al. (2001) performed in Erskine near Missoula, MO.

· 10^{-3} to 10^{-2} log/m for the uncontaminated aquifers from the studies of Flynn (2003), Mallen et al. (2005), and Rossi et al. (1994) performed in Germany and Switzerland,

· 10^{-2} log/m for uncontaminated coarse gravel aquifers from Pang et al. (1998) and Sinton et al. (2000), and 10^{-3} log/m for contaminated coarse gravel aquifers from Noonan and McNabb (1979), Sinton (1980a, 1980b), Sinton and Close (1983), and Sinton et al. (1997), performed in New Zealand.

Consolidated Aquifers

· 10^{-1} to 10° log/m for uncontaminated fractured clay till and fractured clayed shale saprolite from McKay et al. (2000; 1993),

· 10^{-2} to 10^{-1} log/m for uncontaminated fractured gneiss from Champ and Schroeter (1988),

· 10^{-2} log/m for contaminated sandstone from Krapac et al. (2002),

· 10^{-3} to 10^{-2} log/m for contaminated fissured chalk from Beard and Montgomery (1981),

· 10^{-2} to 10^{-1} log/m for contaminated limestone x < 85 m from Krapac et al. (2002) and Mahler et al. (2000),

Table 13. Removal rates of microbes in gravel aquifers estimated from field experiment data.

Reference	Aquifer	Aquifer properties	Contaminant source	V (m/d) or R	x	Microbe	Removal rate λ	r^2 linear	r^2 log	No. of data	Method & data
					m		log/m				
Pieper et al. (1997)	Sand and fine gravel	Cape Cod, θ= 0.39, d=0.6 mm	Clean	0.8–1.3	3.6	PRD1	2.15E-01	0.77	0.86	4	b,m,n
			Sewage	0.4–0.7	3.6	PRD1	2.13E-01	0.43	0.59	4	b,m,n
Harvey et al. (1995)	Sand and fine gravel	Cape Cod. θ = 0.35, d_{50} = 0.59 mm, K = 86.4 m/d, plume 5 km long	Sewage 50 yr	0.3–0.7, R=2–6	3.6	Protozoa	0.43 (0.37–0.50)	0.98–1.0	0.98–1.0	3 ×4	b,h
Harvey et al. (1993)	Sand and fine gravel	Cape Cod, d = 0.50 mm, pH=5.8–5.9, DOC = 1–2 mg/L	Sewage	R=0.8–1.7	6.0	Indigenous bacteria	0.32 (0.24–0.41)			3	a,e
Harvey and Garabedian (1991)	Sand and fine gravel	Cape Cod, d = 0.59 mm, θ=0.35	Sewage	0.34	6.8	Indigenous bacteria	0.15–0.21			2	a,g
Blanford et al. (2005)	Sand and fine gravel	Cape Cod, WT = 3–7 m, K = 60–120 m/d, θ = 0.2–0.4, I = 0.0015, Sewage contaminated zone: pH=6.5, uncont. zone pH = 5.6. Well-sorted, medium-coarse sand with fine gravels	Sewage	0.4, R=0.75–1.7	1.2–7.8	PRD-1	0.89 (0.49–1.11)	linear		31	b,l
			Sewage	0.4, R=0.75–1.7	4.5–11.4	PRD-1	0.11 (0.03–0.18)	linear		21	b,l
			Clean	0.5, =1.02–1.90	1.2–11.4	PRD-1	0.12 (0.06–0.18)	linear		32	b,l
Bales et al. (1995)	Sand and fine gravel	θ = 0.2–0.4, d_{50} = 0.45 mm, K = 60–120 m/d, I = 0.0015, WT = 13.7m asl	Sewage	0.2–0.7	12	PRD1	1.78E-01	0.83	0.88	6	b,e,n
Harvey et al. (1984)	Sand and fine gravel	Cape Cod. θ=0.15–0.19, plume 3.4 km long, sewage plant operated for decades	Sewage		210–970	free-living bacteria	1.10E-03	0.91	0.81	5	b,g
					210–2930	free-living bacteria	6.00E-04	0.92	0.92	6	b,g
Schaub and Sorber (1977)	Silty sands and gravel	Fort Devens, MA, Boston, Ks = 8.64 m/d, infiltration basins operated for over 30 yr.	Sewage		183	Indigenous enteric virus	2.92E-03			2	a,e
					183	f2 phage	1.70E-03			2	a,e
Deborde et al (1998b)	Sand and gravel	Frenchtown High School, near Missoula, θ = 0.2, d = 2.4 mm, I = 0.002, K = 240–300 m/d	Septic tanks Contaminated	Br: 1–2.9	6.6–17.4	MS2	3.92E-01	0.84	0.73	3	b,g
					6.6–17.4	ϕX174 coliphage	1.23E-01	0.91	0.81	3	b,g
					6.6–17.4	background coliphage	1.94E-01	1.00	1.00	3	b,g
Deborde et al. (1998a)	Sand and gravel	Frenchtown High School, near Missoula. Same site as above	Septic tanks	Br: 1–2.9	17.4	male-specific coliphage	2.15E-01	0.98	0.82	4	b,g
			Sewage		17.4	Somatic phages	2.12E-01	0.98	0.79	4	b,g
Deborde et al. (1999)	Sandy gravel	Erskine, near Missoula, θ = 0.15, d = 1.25 (sand) and 12 mm (sand and gravel). WT = 2.1–2.6 m,	Clean	MS2: 23–39	7.5–40.5	MS2	9.94E-02	0.99	0.95	4	b,g
		preferential flow		PRD1: 26–39	7.5–40.5	PRD1	9.47E-02	0.98	0.95	4	b.g
		K = 6,000–13,500 m/d. I=0.00043, V_{Br}=22–30 m/d. pH=7.2		ϕX174: 18–39	7.5–40.5	ϕX174 coliphage	1.51E-01	0.98	0.86	4	b.g
				Polio: 33–45	7.5–19.4	Poliovirus	9.68E-02			2	b.g
Woessner et al. (2001)	Sand and gravel	Erskine, near Missoula, same site as Deborde et al. (1999)	Clean	PRD1: 115	21.5	PRD1	1.21E-02			1	d,h
		θ = 0.15, I=0.00043, K = 36–800 m/d, preferential flow		MS2: 147	21.5	MS2	3.58E-02			1	d,h
		K = 13,000 m/d, V = 27 m/d, V'_{Br} = 129 m/d		ϕx174: 147	21.5	ϕX174 coliphage	5.37E-02			1	d,h
				Polio: 172	21.5	Poliovirus	1.36E-01			1	d,h
Flynn (2003)	Gravel and sand	Munich, Germany, I = 0.005–0.01, K = 432 m/d, WT = 1–3 m, V = 41–90 m/d from uranium, θ =0.25, θ_e = 0.11, d = 0.06–0.6 mm	Tracer	*E. coli*: 50–95	20	*E. coli*	3.30E-03			2	b,e
				p.pitida: 48–101	20	*p. pitida*	4.56E-02			2	d,h
				phage: 53–100	20	H40/1 phage	2.27E-02			2	b,e
Mallen et al. (2005)	Gravel and sand	Munich, Germany, natural gradient, same site as Flynn (2003). Fluvioglacial gravels and sands (90–98% limestone and dolomite detritus).	Tracer	*E. coli*: 48–72	10–20	*E. coli*	3.65E-3–4.33E-3			2	o
				p.pitida: 37–73	10–20	*p. pitida*	2.32E-2–5.57E-3			2	o
				phage: 38–76	10–20	H40/1 phage	2.26E-2–9.87E-3			2	o
Rossi et al. (1994)	Sand and gravel	Postglacial sand and gravel, Switzerland, natural gradient, WT = 1.5–3.5 m, I = 0.004, $V_{micro} = 3.15\,V_{solute}$	Clean	T7: 11–205	14–163	Phage T7	3.74E-02	0.99	0.92	4	b,e,n
			Clean	f1: 11–132	14–163	Phage f1	5.20E-02	0.94	0.78	3	b,e,n

(cont'd)

Table 13. Continued.

Reference	Aquifer	Aquifer properties	Contaminant source	V (m/d) or R	x	Microbe	Removal rate λ	r^2 linear	r^2 log	No. of data	Method & data
					m		log/m				
Sinton et al. (2000)	Coarse gravel	Burnham, New Zealand, θ = 0.2, d_{10} = 0.9 mm, d_{50} = 18.27 mm	Clean	88–112	87	MS2	2.50E-02	0.96	0.99	4	b,e,n
Pang et al. (1998)	Coarse gravel	Burnham, New Zealand	Clean	50–85	87	*B. subtilis* spores	3.12E-02	1.00	1.00	3	b,e,n
Sinton et al. (2000)	Coarse gravel	Burnham, New Zealand	Clean	85–120	21–62	*B. subtilis* spores	4.52E-02			2	b,e,n
		Burnham, New Zealand	Clean	85–120	21–87	*E. coli* J6-2	2.05E-02	0.90	0.85	6	b,e,n
Sinton (1980a)	Coarse gravel	Burnham, New Zealand	Sewage	56–153	920	*E. coli*	4.10E-03	0.67	0.89	4	b,e,n
Sinton (1980b)		Burnham, New Zealand	Sewage	*E. coli* = 203	920	*E. coli*	3.80E-03	0.54	0.71	4	b,e = b,h
Sinton (1980b)		Burnham, New Zealand	Sewage	*B. stearother.* = 188	920	*b. stearothermophilus*	3.40E-03	0.78	0.93	4	b,e = b,h
Sinton and Close (1983)	Coarse gravel	Burnham, New Zealand	Sewage	*E. coli*: 215–265	570	*E. coli* JC 3272	4.15E-03			2	b,g
Noonan and McNabb (1979)	Coarse gravel	Burnham, New Zealand	Sewage	300	920	T4 coliphage	3.80E-03	0.93	0.25	4	b,g
					920	ϕX174 coliphage	1.40E-03	0.86	0.74	4	b,g
Sinton et al. (1997)	Coarse gravel	Templeton, New Zealand, θ = 0.2	Sewage	144–182	445	F-RNA phages	1.80E-03			2	b,e,n
			Sewage	144–182	445	Somatic phages	4.00E-03			2	b,e,n
			Sewage	144–182	445	Faecal coliforms	2.80E-03			2	b,e,n
Sinton and Close (1983)	Coarse gravel	Templeton, New Zealand	Sewage	*E. coli*: 156–190	240	*E. coli* J6-2	9.80E-03	1.00	0.92	3	b,g

10^{-3} log/m for contaminated limestone x = 1250 m from Auckenthaler et al. (2002),

· 10^{-4} log/m for contaminated limestone x = 5000 m from Masciopinto et al. (2008).

The above results suggest that coarse gravel aquifers, chalk aquifers, and karst limestone aquifers have relatively lower capacities for microbial removal, while pumice sand, alluvial sand aquifers, and highly weathered aquifer rocks containing clay have a much greater capacity for microbial removal. However, when contaminant plumes develop over large distances under long-term loading by contaminant sources, even normally capable aquifers can exhaust their capacity for microbial removal, as demonstrated in the removal rates estimated from Harvey et al. (1984). The effect of continuous effluent loading on microbial removal is even showed in experiments performed over very short durations. For example, Wall et al. (2008) observed a progressive reduction in phages mass removal between experiments (93, 75, and 63%) with continuous loading of DOC over 25 d in an 18 cm column filled with pumice sand.

Less Microbial Removal in Contaminated Aquifers, Anoxic Aquifers, and for Waste-Associated Species

Table 13 shows that when a coarse gravel aquifer (Burnham, New Zealand) is contaminated with sewage effluent, its ability in microbial removal is reduced by one order of magnitude (10^{-3} log/m) compared with when it is uncontaminated (10^{-2} log/m). Similarly, a greater removal of PRD-1 is achieved in an uncontaminated zone than in a sewage-contaminated zone as found in a sand and fine gravel aquifer at Cape Cod (Ryan et al., 1999). This is attributed to the influence of sorbed and dissolved organic matter and other anions in the effluent. As organic matter is net negatively charged like microbial particles, they compete with each other for the same sorption sites in the aquifer media and reduce the electrostatic sorption sites available for microbial attachment. Ryan et al. (1999) demonstrated that contaminated aquifer media is more net negatively charged than uncontaminated aquifer media. The above mentioning of "net" negative charge is because both organics and microbial cells can have significant hydrophobic surface loci, which is positively charged (Unc and Goss. 2004).

DeBorde et al. (1998b) found that the removal rates of septic tank waste-associated coliphages (composed of collections of somatic and F-RNA phages) are about twice as slow as MS2 phage and ΦX174 marker phages. A possible explanation for this is that the properties of waste-associated coliphages are much more variable and more resistant to environmental stresses than marker phages cultivated in the laboratory. This finding is consistent with the observation of Jansons et al. (1989) who report that waste-associated viruses penetrate much deeper into the vadose zone than seeded vaccine polioviruses, as mentioned earlier. Other factors could also contribute to this difference, for example, different input configurations (pulse input for the seeded phages vs. continuous input of effluent-viruses) and differential changes in recoverability (viability).

Table 14. Removal rates of microbes in fractured rocks and karst aquifer systems estimated from field experiment data.

Reference	Aquifer	Aquifer properties	Contaminant source	*V*	*x*/depth	Microbe	Removal rate λ	r^2 linear	r^2 log	No.	Method & data
				m/d	m		log/m				
McKay et al. (1993)	Fractured clay till	Ontario. I = 0.24, forced gradient, θ = E-3–E-5 for fracture and 0.32 for clay matrix. K = 2E-10–E-6 m/s. Trench-to-trench constant head tracer test. Vertical transport in saturated zone	Unc.-Tracer	Br: 0.01–0.07	0/5	PRD1	1.00E-01	0.95	0.88	3	b,g
				Phage: 2–>5	0/5	MS2	1.30E-01	0.97	0.84	3	b,g
					4/1.6–2.6	PRD1	0.82 (0.43–1.03)			3	a,g
					4/1.6–2.6	MS2	1.28 (1.02–1.45)			3	a,g
McKay et al. (2000)	Fractured clayey shale saprolite	Highly weathered, natural gradient, I = 0.04, K = 13.3 cm/h, θ = 0.12–0.52, phages moved 500 times faster than dye tracer	Unc.-Tracer	PRD: 11–56	18	PRD1	3.00E-01	linear		3	b,k
				MS2: 11–42	18	MS2	3.31E-01	linear		3	b,k
Champ and Schroeter (1988)	Fractured gneiss	Chalk River, Canada, natural and forced gradients	Tracer	*E. coli*: 6–7	12.7	*E. coli*	0.087–0.143			2	c, i
Beard and Montgomery (1981)	Fissured chalk	Alresford, Hampshire, UK, sewage discharge through French drainage and soakage operation systems, WT = 26–35 m bgl, including vadose zone	Sewage	43–350	50–450	Fecal coliforms	3.70E-03	0.92	0.81	4	b,e
					40–360	Fecal coliforms	8.40E-03	1.00	0.95	4	b,e
					425	Fecal coliforms	4.6E-2(1.2E-2–6.7E-2)			8	d,h
Auckenthaler et al. (2002)	Karst limestone	Switzerland, highly porous limestone embedded in low permeable formation, natural gradient, including vadose zone	Tracer	Phage: 846	1250	H4 phage	1.28E-03			1	d,h
				Solute: 770	1250	H40 phage	1.28E-03			1	d,h
Mahler et al. (2000)	Karst limestone	K = E-7–E-9 m/s, slight fractured zone	Creek		10	Enterococci	2.15E-01			2	a,f
					10	Fecal coliform	5.23E-02			2	a,f
		K = E-1–E-3 m/s, highly fractured zone	Creek		9	Enterococci	1.32E-01			2	a,f
					9	Fecal coliform	6.69E-02			2	a,f
Masciopinto et al. (2008)	Fractured Limestone	Italy, WT = 32 m, wastewater injection, limestone formations are significantly fractured and very permeable. θ_e = 0.0032, model simulated removal rate 0.1–0.9 d^{-1}.	Sinkhole Contaminated	50 (10–250)	320–5000	Somatic phages	8.00E-04	0.80	0.92	5	b,g
					320–5000	*Clostridium*	4.00E-04	0.81	0.86	5	b,g
					320–5000	Streptococci	9.00E-04	0.93	0.76	5	b,g
					320–5000	Fecal coliforms	9.00E-04	0.81	0.95	5	b,g
					320–5000	*E. coli*	1.00E-03	0.95	0.94	5	b,g
Krapac et al. (2002)	Limestone	Underlain by silt-clayey silt loess and diamicton, and shale	Pig manure pit run for 4 yr		41–85	Fecal streptococcus	1.6E-2(1.5E-2–1.7E-2)			2	a,f
	Sandstone	Underlain by 5 m clay diamicton	Pig manure pit run for 4 yr		24–44	Fecal streptococcus	4.13E-02	0.98	0.96	3	b,f

Table 15. Removal rates of microbes in sand and gravel aquifers derived from riverbank filtration monitoring

Distance from river	Microbe	Removal rate λ	No.	Method & data
m		log/m		
Medema et al. (2000), River Meuse, the Netherlands, pumping rate 140 m³/h, sandy gravel				
135 (15–150)	Somatic Coliphages	2.96E-02	3	d,h
135 (15–150)	SSRC	1.04E-02	3	d,h
135 (15–150)	Coliform	1.56E-02	3	d,h
15–25	F-RNA phages	1.67E-01	2	a,j
0–15	All of the above and reovirus	2.40E-1–2.67E-1	5	d,h
Wang et al. (2000), Ohio River at Rier Mile 592, Louisville, V = 7.32–8.78 m/d				
15	Aerobic spores	8.00E-02	4	b,k
30.5	Fecal coliform	1.02E-1(3.1E-2–1.48E-1)		a,e
Wang (2002), Ohio Mile 592, Kentucky, watertable 120 m above sea level, θ = 0.4, V = 3–4 m/d				
30	Protozoa	7.00E-2(4.00E-2–9.00E-2)	3	a,e
30	Aerobic spores	1.7E-1(1.2E-1–2.7E-1)	3	a,e
Weiss et al. (2003), Ohio River Indiana-America Water Company at Jefferson, IN				
V = 9.33–13.6 m/d. θ = 0.2–0.3, gravel mixed with fine-media sand				
30	F-RNA phages	7.00E-02	2	d,h
27	Clostridium	1.26E-01	2	d,h
Weiss et al. (2005), Ohio River Indiana-America Water Company at Jefferson, IN				
30	*Bacillus*	7.00E-02	6	a,e
177	*Bacillus*	1.50E-02	9	a,e
177	Somatic bacteriophage	1.80E-02	9	a,e
Weiss et al. (2005), Wabash River Indiana-America Water Company at Terre Haute, IN				
27	*Bacillus*	8.10E-02	9	a,e
27	*Clostridium*	1.50E-02	9	a,e
122	*Clostridium*	1.90E-02	5	a,e
Weiss et al. (2005), Missouri River-America Water Company at Parkville, MO				
37	*Bacillus*	2.20E-02	9	a,e
37	Total coliforms	1.65E-01	9	a,e
37	F-RNA phages†	6.80E-02	9	a,e
37	*Bacillus*	7.00E-02	4	a,e
37	Total coliforms	1.49E-01	4	a,e
37	F-RNA phages†	5.70E-02	4	a,e

† In original paper it is noted as a male-specific bacteriophage.

Anoxic groundwater systems have oxygen and nitrate concentrations below 0.5 mg/L (Schijven and Hassanizadeh, 2002). van der Wielen et al. (2006; 2008) reported that the removal rates of phages MS2 and ΦX174 in anoxic aquifers were considerably lower than their previously reported removal rates in oxic aquifers due to lower inactivation and adsorption rates under anoxic conditions. Tate (1978) also reported that the greatest coliform survival in soils was seen under anaerobic conditions. Oxygen is a major regulator of microbial survival (Roslev et al., 2004). Under oxic conditions, the oxidation of lipids can alterate membrane structure and function, and thus damage microbial proteins (Kreier, 2002). Furthermore, metal oxides are present in the oxidized conditions, which would also enhance microbial removal.

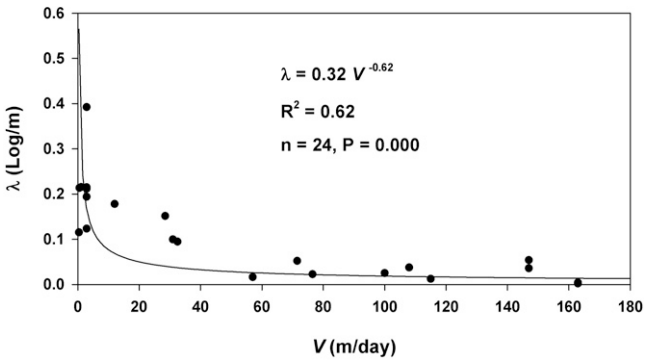


Fig. 1. Correlation between spatial removal rate and pore-water velocity for phage transport in gravel aquifers.

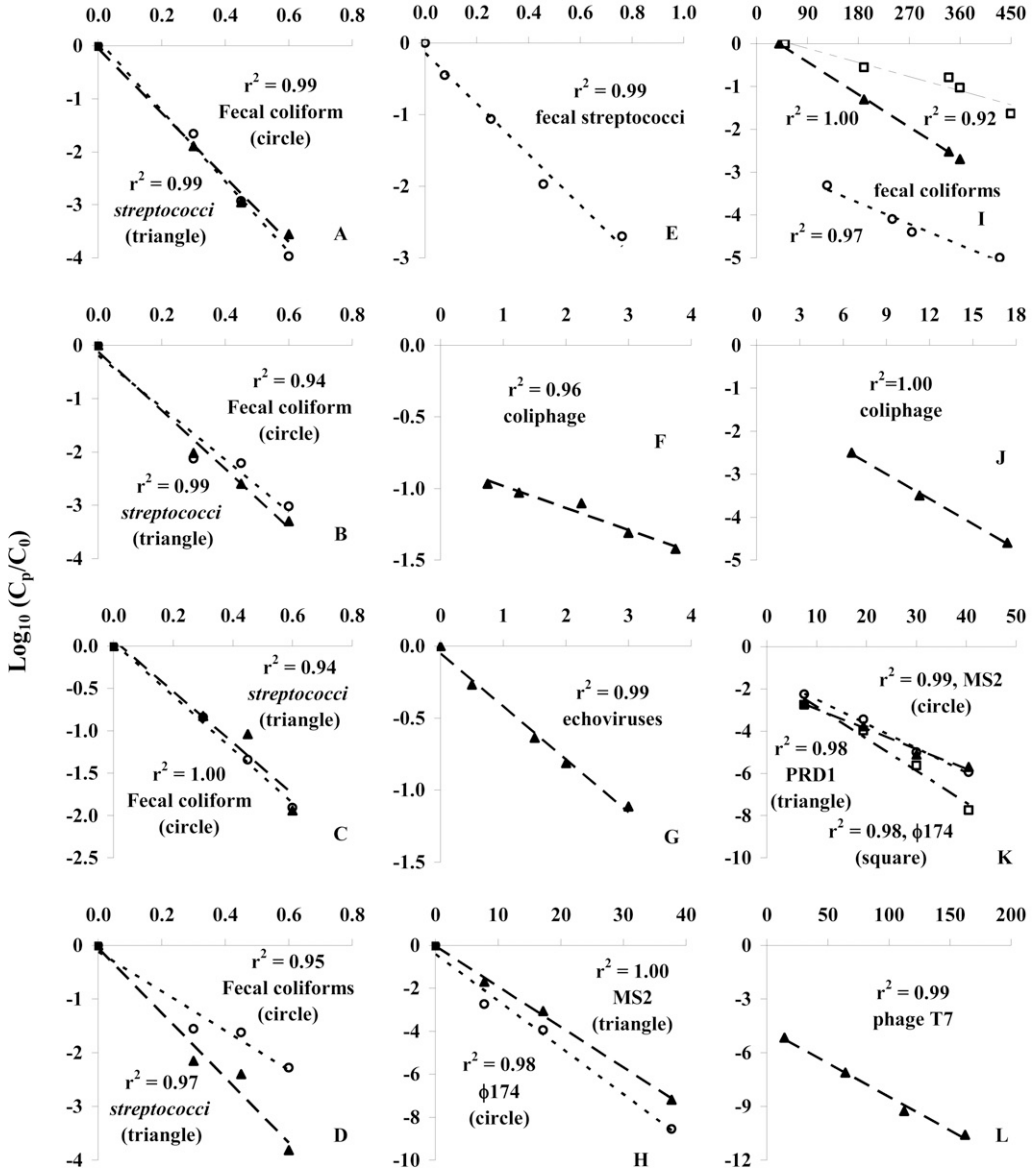
Effect of Pore-Water Velocity on Microbial Removal Rates

Of all of the aquifer properties (distance, porosity, particle size), the parameter that shows a clearest correlation with removal rates is pore-water velocity, which would also be corrected with connected porosity and pore-size distribution. Selecting removal rates derived from gravel aquifers for phages, Fig. 1 demonstrates that the removal rate decreases with pore-water velocity. This is consistent with the findings from soils and vadose zones – that removal rates decrease with increases in hydraulic conductivity, hydraulic loading, and infiltration rates. It is important to note that the exponent of the power function fitted to the λ vs. V plot is 0.62 rather than unity as expected from Eq. [3], $\lambda = kV^{-1}$. This is probably an artifact caused by the mixture of pore-water velocities for microbial tracers and conservative tracers used in the plot. This is because microbial pore-velocities were given in some studies but not in other studies (Table 13). Pore-size exclusion in heterogeneous large-pore aquifers and retardation in low-flow aquifers could lead to the velocities of microbial tracers being quite different from those of conservative solute tracers. Nevertheless, Fig. 1 illustrates the inverse relationship between removal rate and pore-velocity.

Removal With Distance: Log (C_p/C_0) – x Functions

In addition to fitting the log (C_p/C_o)– x data with the linear functions, log functions are also fitted to the data and their r^2 values are listed in Tables 6, 8 to 9, and 11 through 14. It should be noted that these r^2 values only provide a relative comparison in this paper, regardless of whether they are statistically significant in relation to the number of datasets analyzed. Comparing the r^2 values for a total of 87 comparable cases, 70% of cases fit better with the linear functions and 30% of cases fit better with the log functions. Both relationships are shown at different transport scales. Selected examples for linear and log functions are shown in Fig. 2 and 3, respectively.

When fitting a function to the log (C_p/C_0)– x data, caution is advised when considering whether to include the origin, x = 0. For soils and vadose zones, the concentration of the injection solution is often the actual input concentration to the systems at x = 0 (C_0). However, for groundwater studies, there is some uncertainty about C_0 in aquifer tests where the initial



Note			
	A	Soil	Karathanasis et al. (2006), loamy sand, contaminated
	B	Soil	Karathanasis et al. (2006), loamy sand, contaminated
	C	Soil	Karathanasis et al. (2006), sandy loam, contaminated
	D	Soil	Karathanasis et al. (2006), clay, contaminated
	E	Soil	Schaub & Sorber (1977), silty sand and gravel, contaminated
	F	Vadose zone	Ho et al. (1992), sand contaminated
	G	Vadose zone	Jansons et al. (1989), sand, contaminated
	H	Aquifer	van der Wielen et al. (2008), coarse sand, uncontaminated
	I	Aquifer	Beards & Montgomery (1981), fissured chalk, contaminated
	J	Aquifer	Deborde et al. (1998b), sand and gravel, contaminated
	K	Aquifer	Deborde et al. (1999), sandy gravel, uncontaminated
	L	Aquifer	Rossi et al. (1994), sand and gravel, uncontaminated

Fig. 2. Selected examples of linear function between $Log_{10}(C_p/C_0)$ vs. transport distance.

solution is injected into a test well. The concentration of the injection solution could be much greater than the actual input concentration of microbial tracers in the groundwater at $x = 0$ due to dilution of the injection solution in the well.

If $x = 0$ is included in the log (C_p/C_0)– x plot when the value for C_0 in the injection solution is much higher than the actual concentration at $x = 0$, it will result in an artificially high removal rate, especially for the distances near the sources. The single point at $x =$

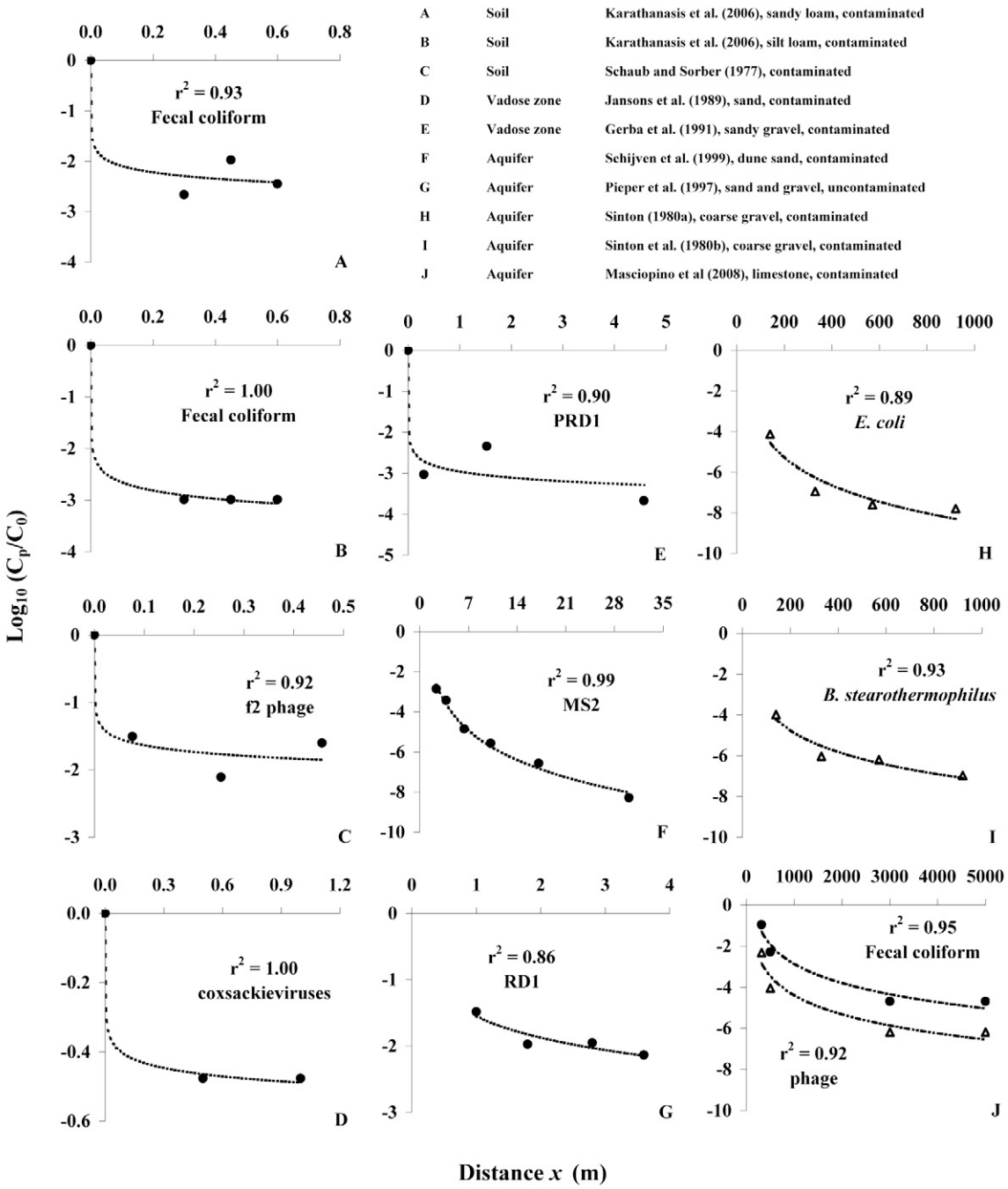


Fig. 3. Selected examples of log function between $\text{Log}_{10}(C_p/C_0)$ vs. transport distance.

0 would dominate the shape of the log (C_p/C_0)– x plot. Fortunately, when $x = 0$ is excluded, the value of C_0 has no impact on the derivation of the removal rate, which only depends on the slope of the log (C_p/C_0)– x plot, not the intercept. Whether $x = 0$ is excluded depends on how much the injection solution is diluted in the injection well. For example, where a very small volume of a high concentration solution is injected into a large segment of a well, it is best to exclude $x = 0$ from the data analysis; but where a large amount of tracer solution is injected into a small well, $x = 0$ could be included.

The linear log (C_p/C_0)– x relationship implies that the concentration of mobile microbial tracers decreases exponentially with transport distance at a constant first-order removal rate, which is consistent with the concept of conventional transport models and filtration theory (Logan et al., 1995; Yao et al., 1971; Matthess et al., 1988). In contrast, the log log (C_p/C_0)– x relationship suggests that the concentration of mobile microbial particles decreases with transport distance in a power function and that the microbial removal rate decreases with distance (or hyper-exponentially), which contradicts the concept of conventional transport models and filtration theory.

As summarized in the reviews by Bradford et al. (2006a) and Johnson et al. (2007), the deviation of observed colloid concen-

trations from the predictions of filtration theory has been observed in many laboratory studies. These studies typically involve the use of uniform fine-grain porous media (predominantly sand), slow flow rates, and small columns (in a transport scale of centimetres). Schijven and Hassanizadeh (2000) in their review of some laboratory and field studies performed at Cape Cod (a transport scale of up to 4 m), also indicated initially higher virus removal rates with distance. However they acknowledged that this phenomenon was not observed by DeBorde et al. (1998b) in their field study, and removal rates for somatic phages and FRNA phages were linear over a distance of 18 m. Note that the sand and gravel aquifer media investigated in DeBorde et al. (1998b) was much coarser (d = 2.4 mm, Table 13) in comparison with that in Cape Cod (d = 0.45–0.60 mm, Table 13).

Various explanations have been proposed in the literature for the observed discrepancies in filtration theory predictions, but they can be placed in three categories: (a) the presence of straining, (b) heterogeneous attachment conditions, and (c) unfavorable attachment conditions. These three categories are explained next.

Distance-dependant straining processes, which differ from attachment, could be largely responsible for the observed discrepancies for bacteria, protozoa, and large-sized colloids in fine porous media (Bradford et al., 2003, 2004, 2006b; Foppen et al., 2005; Tufenkji et al., 2004). Although straining usually applies to bacteria and protozoa, it may also apply to viruses that are bound to large colloidal particles. The deviations from predictions from filtration theory tend to increase for larger colloids and finer textured porous media (Bradford et al., 2003; Tufenkji and Elimelech., 2005). Straining occurs when particles become trapped in pore throats that are narrower than the particle diameter. Physical straining is of less importance as a deposition mechanism than attachment, especially for viruses. Straining is however significant when the ratio of the colloid to media grain diameter is >8% (McDowell-Boyer et al., 1986), and will occur when the ratio is >0.5% (Bradford et al., 2004). Microbial removal by straining can be evaluated using the model developed by Bradford et al. (2003). Straining is a function of the size of colloids and grains, pore space geometry, features of colloid and solid surface, solution chemistry, system hydrodynamics, and the colloid concentration (Bradford and Torkzaban, 2007; Bradford et al., 2006c). Hydrodynamic shear can diminish straining at higher velocities and/or slowly mobilize strained colloids down gradient (Bradford et al., 2006c). Bradford et al. (2006c) have experimentally demonstrated that hyperexponential deposition profiles are typically associated with finer textured sands and lower flow rates with significant straining, which occurs primarily at the inlet of the columns. In contrast, they observed uniform deposition profiles and gradually decreasing concentrations with depth in coarser textured sands and at higher Darcy velocities.

The observed discrepancies from filtration theory could be also due to the fact that the filtration theory was derived from ideal attachment conditions, for example, clean beds with uniform physical and chemical properties, colloids with uniform properties, and constancy in chemical solution properties. Wide varieties of attachment conditions can occur during colloid transport, for example, heterogeneity in microbial properties (type, size, density, charge, survival characteristics, strains, isolates, and aggregation with colloids) and porous media (Schijven and Hassanizadeh, 2000), surface heterogeneity (e.g., charge and roughness) of colloids and/or porous media (Kretzschmar et al., 1997; Redman et al., 2001), and colloid detachment (Tufenkji et al., 2003). These non-uniform attachment conditions may also result in hyperexponential or nonmonotonic concentration profiles, as observed in the studies described above. In addition, changes in the redox conditions of mineral surfaces, which commonly occur in geochemically heterogeneous subsurface media, could also significantly change the attachment behaviors of microbial contaminants. The effect of redox conditions on microbial removal is clearly demonstrated in the field by Schijven et al. (2000). This study showed a very strong nonlinear log-removal of phages and spores over distance. Their interpretation was that this is probably due to the preferable attachment of microbes to patches of ferric oxyhydroxides that are present within 8 m of the injection point, but not thereafter.

Unfavorable attachment conditions, referring to interactions when a colloidal particle and its collector are like-charged, are also interpreted as a possible reason for the observed discrepancies from filtration theory due to the presence of a repulsive energy barrier between the colloid and the mineral surface (Johnson et al., 2007; Li and Johnson, 2005; Tufenkji and Elimelech, 2004). Unfavorable attachment conditions typically occur in the presence of organic matter as both colloids and organic matter are net negatively charged in most natural environments. Thus it is expected that microbial transport in sewage-contaminated subsurface media will be under unfavorable attachment conditions. In contrast, under favorable attachment conditions, for example, in the absence of repulsive interaction energy barriers, the observed concentrations of mobile and retained colloids decrease exponentially with distance from the source (Li et al., 2004, 2005; Tufenkji and Elimelech, 2004; Johnson et al., 2007), which is consistent with filtration theory.

Whether a Log (C_p/C_0)– x function is linear or higher order (e.g., two rate models or power law) could be determined by examining whether the difference in the slope of Log (C_p/C_0)–x plot is statistically significant using the likelihood ratio tests (Cox and Hinkley, 1974). A likelihood-ratio test is a statistical test for making a decision between two hypotheses based on the value of this ratio. The detailed description of this method and its application for analyzing experimental data is given in Schijven et al. (2002, 2004).

Log Log (C_p/C_0) – x Functions

The r^2 values shown in Tables 6, 8 to 9 and 11 through 14 suggest that log functions are superior to linear function for describing some of the data from DeBorde et al. (1998b), Gerba et al. (1991), Karathanasis et al. (2006), Masciopinto et al. (2008), Pieper et al. (1997), Schijven et al. (1999), and Sinton (1980a, 1980b). Data obtained from the contaminated zone by Blanford et al. (2005) could in fact be better fitted with log functions (not shown in this study) although two linear fits were applied in their original paper.

The plausible explanations for these observed discrepancies from the predictions of conventional models and filtration

theory are due to (1) the presence of organic matter, (2) kinetic detachment, (3) variable microbial populations, (4) change in ionic strength, and (5) straining. These are explained below.

All of the field studies described above were under the influence of organic matter. The net negatively charged dissolved and colloidal organic matter in the effluent would compete with microbial particles for attachment to solid phase (Powelson et al., 1991; Schijven and Hassanizadeh, 2000), thus fewer attachment sites in the solid phase would be available for microbial particles. Meanwhile, microbial particles in the effluent could be adsorbed onto organic colloids (Sobsey et al., 1991) and co-transported with mobile organic colloids. The adsorption of microbial particles onto organic colloids in the effluent may protect them from inactivation (Alley, 1993; Canter et al., 1987). As a result of repulsive energy barrier and reduced inactivation, microbial particles would travel greater distances. On the other hand, when microbial particles are attached to immobile organic colloids, or when their associated mobile organic colloids are adsorbed to solid phase and become immobile, the presence of organic matter would inhibit transport of microbial particles. This dual role of organic colloids in facilitating and inhibiting transport of other contaminants is experimentally demonstrated in Totsche et al. (1997), and this is also expected to be true for microbial contaminants. If the fraction of microbial particles bound with immobile organic colloids reduces over distance due to dilution of groundwater, removal rates could then decline with distances.

Microbial detachment could also contribute to the reduction of removal rates over distance, resulting in concentrations to change little at larger distances after the initial exponential decline. This is shown in some of the field studies performed at Cape Cod. For example, Pieper et al. (1997) found that after a travel distance of 3.6 m, an almost constant amount of PRD1 continued to break through. Harvey et al. (1984) observed little variation in bacterial levels in contaminated groundwater samples taken beyond a transport distance of 1 km. Blanford et al. (2005) observed that after first 1 m of transport in the uncontaminated zone and 4 m in the contaminated zone, PRD-1 levels reduced very little and these levels remained almost constant in both zones for the remaining travel distances (up to 13 m), irrespective of variations in geochemical properties within and between the two zones. It is expected that detachment would be greater in contaminated aquifers than in uncontaminated aquifers as dissolved organic matter could detach microbial particles that are bound to the mineral surfaces of aquifer media, a mechanism used to elute viruses adsorbed onto membranes (Lytle and Routson, 1995).

Populations of a particular microbial species in sewage effluent can vary widely in terms of differences in size, buoyant density, charge, and survival characteristics (Jansons et al., 1989; Sharma et al., 2000). Microbe subpopulations which are larger, heavier, less negatively charged, and shorter-lived will be removed at shorter distances, while smaller, lighter, more negatively charged, and longer-lived microbial particles will travel further. The effect of microbial cell size and buoyant density on transport distances has been demonstrated in Harvey et al. (1997) and Harvey et al. (2002). Their studies suggest that laboratory-grown bacteria are much larger and heavier than free-living bacteria in the sewage plume, thus could be removed at shorter distances. In contrast, the free-living bacteria vary widely in sizes and their densities, thus could be removed at different transport distances. The change in microbial populations, thus their properties, with distance will also reduce removal rates over distance.

Dilution of the input effluent by groundwater would reduce the ionic strength of the solution over the transport distance. This will lower the deposition of microbial particles over distance as colloid deposition is positively correlated to the ionic strength of the solution (Bradford et al., 2007).

Some field data described above are derived from sandy media where flow velocities are generally low. Hence, physical straining, which primarily occurs near contaminant sources, could be significant. Even if the ratio of the microbial particles themselves to media grain do not meet the criteria of 0.5% for straining, the microbial particles (possibly even viruses) could be strained out with the larger colloids, as most microbial particles are associated with colloids in the wastewater and septic tank effluent (Sobsey et al., 1991). Straining is however expected to be insignificant in coarse gravel and karst limestone aquifers as microbial particles travel largely through preferential flow paths. Thus the log-log functions displayed in these aquifers (Fig. 3) is believed to be largely due to nonuniform attachment conditions, particularly the change in microbial populations (their sizes, buoyant density, charges, survival characteristics, etc.) and detachment, as discussed earlier.

In the studies described above, it is often the case that for the same experiment, the log functions are better fitted to some microbial species while linear functions are better for other species. This supports the hypothesis that heterogeneity among microbial particles themselves (type, size, density, charge, strains, survival characteristics, isolates, and aggregation with colloids) may also affect their deposition profiles (Schijven and Hassanizadeh, 2000).

Linear Log(C_p/C_0) – x Functions

The r^2 values listed in Tables 6, 8 to 9 and 11 through 14 suggest that the majority of field data (70%) are better fitted with the linear-log functions, which applies for both contaminated and uncontaminated media at various transport scales. This function is also demonstrated in the original papers by Blanford et al. (2005), DeBorde et al. (1998b), McKay et al. (2000), Richardson and Rusch (2005), Wang (2002), and Wang et al. (2000), especially when the origin $x = 0$ is excluded.

In heterogeneous media microbial transport occurs primarily through continuous large pores (macropores) and preferential flow paths (Jamieson et al., 2002; Pang et al., 2008; Wollum and Cassel, 1978), where flow velocities are the highest. In such an environment, microbial transport approximates a piston flow albeit with a much reduced effective porosity (Germann et al., 1987), thus distance-dependent straining processes are expected to be minimal, particularly if microbial particles have traveled over long distances. When microbial transport occurs primarily through macropores and preferential flow, microbial detachment is often negligible, especially in uncontaminated media (Pang et al., 2008). For uncontaminated aquifers where there is no straining, attachment and inactivation (both being first-order processes), will be

the predominant mechanisms for microbial removal, thus the first-order law assumed in the classic transport models and filtration theory is appropriate for describing microbial removal. In uncontaminated large-pored media where there is no straining, the linear-log relationship is evident even on a laboratory scale, for example, in an 8 m long column filled with pea gravel (Close et al., 2006). The linear-log relationship is shown even during microbial transport though 30-cm long intact soil cores comprising uncontaminated clay soils (Guimaraes et al., 1997).

For tracer experiments conducted in uncontaminated subsurface media, the linear-log functions are commonly seen. Compared to organically contaminated media, uncontaminated subsurface media contain much less organic matter, thus less repulsive force between microbial particles and the surface of porous media. This would result in a greater microbial attachment, which is essentially irreversible in uncontaminated soils (Pang et al., 2008). Unlike microbial contaminants associated in sewage effluent, laboratory strains of a microbial tracer are much uniform in size, shape, density charge and survival rate, thus they would travel at the same speed, yielding a constant removal rate.

Although the linear-log relationship tends to occur more frequently in uncontaminated aquifers, it is also evident in effluent-contaminated aquifers in this study. When preferential flow controls microbial transport, its effect may override the influence of unfavorable and heterogeneous attachment conditions that are typically associated with effluent. Within preferential flow paths, effluent dilution by groundwater is relatively rapid, causing a quick reduction in the concentration of organic matter with distance. This lessens the repulsive energy generated from the net negatively charged organic matter, prompting the attachment of microbial particles onto the aquifer media. This may balance the effect of ionic strength on deposition. As mentioned earlier, colloid deposition will reduce with decreased ionic strength as a result of dilution.

The fact that the majority of field data showed a linear $\log(C_p/C_0)$- x relationship suggests that the REV concept is valid for microbial removal rates. When the origin or the first few meters of a $\log(C_p/C_0)$- x plot is/are excluded, the transport scale is large enough for $x \geq$ REV thus the effect of heterogeneity becomes less important so the removal rate can be averaged into a single representative value of statistical and physical significance. When $x <$ REV, the removal rate cannot be well defined (especially with the artifact from the dilution of the injection solution in groundwater) and the aquifer media cannot be treated as a continuum to yield a value that is representative of the whole. The REV concept may also explain why the removal rates derived from laboratory columns (typically $x <$ REV) are often orders of magnitude higher than those obtained from field conditions for the same aquifer media.

Implications of Removal Rates for Estimating Setback Distances

Methods

The removal rates provided in this study have some important implications in relation to setback distance estimations. They can be used in transport modeling after conversion to temporal removal rates if pore-water velocities are known (see Eq. [3]). For people who have little knowledge of transport modeling, the simple method given below could be used to roughly estimate a minimum setback distance.

Assuming a continuous constant input of effluent, ignoring dispersion, the total reduction in microbial concentrations at a steady state, can be calculated from the formula (based on the concepts of Eq. [4] and [5])

$$n = ST + H_f \lambda_f + H_s \lambda_s + H_v \lambda_v + L_a \lambda_a \quad [8]$$

where, n is the total $\log_{10}$ reduction of microbial concentration between the contaminant source and receiving water, ST is the log reduction of the microbial concentration in the on-site treatment system itself, H is the thickness or vertical distance (m), L is the horizontal distance (m), and λ is the removal rate (log/m). The subscripts f, s, v, and a are for the backfill material in the disposal system (trench or basin), soil of the drainage field, vadose zone, and aquifer, respectively. The target level for total microbial reduction (n) depends on the purpose of the receiving water and the initial concentrations of pathogens in the effluent.

For drinking water, the maximum allowable value (MAV) used in the Netherlands is 2×10^{-7} viruses/L (Schijven and Hassanizadeh 2002; Schijven et al., 2006) to minimize the risk of infection to less than10^{-4}/person/yr, as estimated from dose-response relation for rotaviruses (Regli et al., 1991). Based on the same approach, the estimated MAV for protozoa is 6.75×10^{-7} *Giardia*/L (Regli et al., 1991). The criterion of 2×10^{-7} viruses/L is also adopted in USEPA guidelines (USEPA, 1992). The MAV used in New Zealand for fecal bacteria is less than 1 *E. coli*/100 mL (Ministry of Health, 2005). The guideline value used for shellfish-growing waters in the United States is less than 4×10^{-2} enteric viruses/L (Kohn et al., 1993). In New Zealand recreational surface water bodies should contain less than 126 *E. coli*/100 mL (Ministry for the Environment, 1999).

Information on pathogen concentrations in human effluent is largely available for centralized sewage treatment plants. The survey data of Greening et al. (2000) and Lodder and de Roda Husman (2005) suggest that the concentration of enteroviruses in centralized systems is typically in the order of 10^2 pfu/L in raw effluent and 10^1 pfu/L in treated effluent. However, during disease outbreaks enteroviruses levels can be as high as 10^5 pfu/L in raw effluent and 10^4 pfu/L in treated effluent as shown in the survey data from Dahling et al. (1989). Pathogen concentrations in individual septic tanks are expected to vary much more widely compared with homogenized effluent in centralised treatment systems, because the concentrations will depend on whether infected people are living in the dwellings.

If the values of other components are known, the horizontal setback distance can be calculated from

$$L_a = \frac{n - (ST + H_f \lambda_f + H_s \lambda_s + H_v \lambda_v)}{\lambda_a} \quad [9]$$

Readers should incorporate some uncertainty into their estimations by considering pathogen concentrations for typical and outbreak

situations, the lowest and average removal rates for the subsurface media of their concern, and other specific factors of interest.

If the average velocity of microbial travel (V) through each media is known, the total travel time of the microbial contaminant, $T_{\text{total travel}}$, can be estimated from

$$T_{total\,travel} = t_{ST} + t_f + t_s + t_v + t_a = t_{ST} + \frac{H_f}{V_f} + \frac{H_s}{V_s} + \frac{H_v}{V_v} + \frac{L_a}{V_a} \quad [10]$$

where t is the resident time of the microbial contaminant in each media, and t_{ST} is the setting time of the effluent within the treatment system.

Limitations and Warnings

As mentioned above, Eq. [9] does not consider dispersion. For soils, vadose zones, and sand aquifers, dispersion is generally small but for heterogenous aquifers, it could be significant. As illustrated in Fig. 4, with an increase in dispersion, microbes travel further hence the distance required for a specific log-reduction is greater. Thus the horizontal distances estimated from the above formula could be much underestimated. However, the spatial removal rates determined from field-observed concentrations have already incorporated with the effect of dispersion. This may offset some errors generated by using the above simplified formula.

As discussed previously, removal rates are specific to the physical and chemical properties of microbes (type, size, density, charge, strains, isolates) and the subsurface media (macropores or preferential flow paths, flow rate, hydraulic loading rate, porosity, lithology, sorbed organic matter), solution chemistry (effluent characteristics, dissolved organic carbon, pH, ionic strength, colloidal concentration), transport scale, and the duration of contamination. Readers should try to best match all experimental and environmental conditions when choosing a removal rate from the database provided. However, the most important issue is to match the flow rate as removal rates are most closely related to pore-water velocity. The accuracy of the estimation of a specific setback distance using the removal rates determined from this study will depend on the similarities between the conditions associated with the specific system and those conditions that are associated with the derived removal rate presented here.

During the estimation of setback distances, caution should be exercised when extrapolating distances to those beyond the transport scales from which the removal rates are derived. As mentioned earlier, removal rates may not be constant and may slow down with distance. This is particularly relevant for fine grain aquifers and aquifers under a long-term continuous input of effluent. However, even with a hyper-exponential concentration profile, the linear relationship can be applied for certain ranges of distance.

It should be pointed out that the removal rates determined from experiments with point sources injected with pulses of microbial solutions are expected to be higher than those determined from area sources under long-term contaminant loading. This is because (i) field-measured concentrations have already accounted for the effect of dispersion, which is expected to be greater for the concentrations measured down-gradient of point sources; and (ii) the capacity of subsurface media in attenuating microbial contaminants deteriorates with continuous contaminant loading as more and more attachment sites are occupied by organic matter, as discussed earlier. Thus removal rates determined from area sources over long-term effluent loadings are recommended for use as a conservative approach.

Summary

In this study, spatial removal rates of microbial contaminants in subsurface media were estimated by analyzing a large body of published data obtained from field experiments and large intact soil cores. The removal rates have assumed all irreversible processes. When there are sufficient sampling locations, a λ value is interpreted from the slope of log (C_p/C_o) vs. x plot. When numbers of sampling locations are limited, a λ value is directly calculated from the peak-concentration, or recovery efficiency (or mass recovery), or converted from the temporal rate rate (attachment rate + inactivation rate) if modeling results are available. For groundwater tracer experiments, the origin $x = 0$ is often excluded from the slope to avoid the error in the calculation of C_o due to the dilution of the injected solution in groundwater.

The patterns and magnitudes of removal rates for microbial transport in a wide range of soil, vadose zones, and aquifer media have been identified and are summarized in Table 16. The results of this study suggest that the subsurface media that are most effective at microbial removal, thus suitable for effluent land disposal and would require smaller setback distances, are allophanic soils, pumice sand, fine sand, and highly weathered aquifer rocks, while the least effective subsurface media are structured clayey soils, stony soils, coarse gravel aquifers, fractured rocks, and karst limestones. Clay particles are very effective at filtering microbial contaminants under ideal matrix flow conditions, but clay soils under field conditions are susceptible to shrinking and cracking, often lowering removal rates in comparison with sandy soils. Removal rates are more variable in soils containing clay and gravels than in fine textured soils and volcanic soils. Removal rates show a clear inverse relationship with pore-water velocity, hydraulic conductivity, hydraulic loading, and infiltration rates. By using appropriate management practices, the efficiency of subsurface media for microbial removal could improve. Reducing irrigation rates, periodic drying and wetting, disturbing macropores in soils using tillage, especially in vegetated soils, could achieve this.

For the same media, the removal rates for viruses are in the same order of magnitude as they are for bacteria, and can be lower or higher (due to possible removal with associated large colloids). Removal rates are lower for enteroviruses than for other human viruses; for MS2 phage than for other phage species (except for PRD-1 sometimes); for waste-associated microbial species than those cultivated in the laboratory; and for contaminated aquifers than for uncontaminated aquifers. For relatively homogeneous sand and fine gravel aquifers, removal rates remain within the same order of magnitude for both uncontaminated and contaminated conditions, but they are one order of magnitude lower for heterogeneous coarse gravel aquifers when they are derived from contaminated conditions.

Both linear and log functions of log (C_p/C_o) vs. x plots are displayed for the field data obtained from various transport scales,

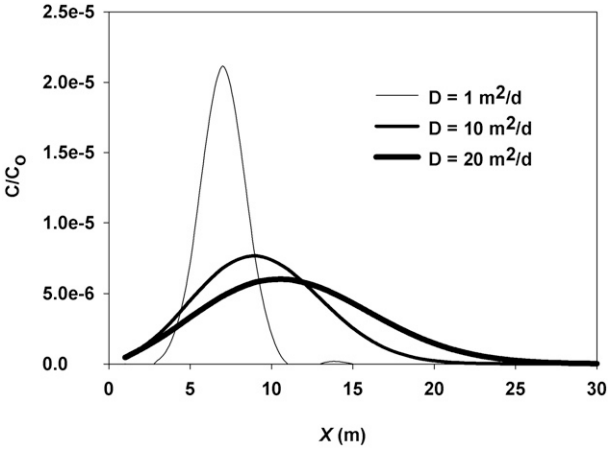


Fig. 4. Hypothetical example of the effect of dispersion on microbial transport distance as simulated from CXTFIT.

with 70% (61 out of 87 pairs) better fitted with the linear functions. The linear function implies that microbial removal is a first-order process and removal rate is constant with distance (when $x \geq$ REV), which is consistent with the assumption made in conventional transport models and filtration theory. In contrast, the log function suggests that microbial removal follows a power law and removal rate declines with increasing distance, contradicting the conventional transport models and filtration theory. For the same experiment, the best fit functions often change with different microbial species. Data that show the log patterns are predominately derived from contaminated media, especially in relatively fine aquifer media. Unfavorable attachment conditions due to the presence of organic matter, hetergenious attachment conditions (due to heterogeneity in the properties of microbial contaminants, change in solution chemistry, detachment), and physical straining (especially when microbial particles are associated with colloids in effluent) may have caused the discrepancies from the linear pattern predicted from traditional transport models and filtration theory. The linear pattern is seen for data obtained from both uncontaminated and contaminated media. In heterogeneous aquifer and vadose zone media and structured soils, microbial transport occurs primarily through continuous large pores and preferential flow paths, and almost follows a piston flow with much reduced effective porosity. Under such conditions, detachment and distance-dependent straining processes are expected to be minimal, and the effect of preferential flow on microbial transport may override the influence of unfavorable and heterogeneous attachment conditions that are typically associated with effluent.

Despite the limitations due to the assumptions and simplifications used in removal rate estimations, the results of this study provide useful information on the relative abilities of subsurface media in removing microbial contaminants. The results of this study have important implications for the determination of safe setback distances. Removal rates are specific to the physical and chemical properties of microbial contaminants and subsurface media, solution chemistry, transport scale, the type of contaminant source, and the duration of contamination. Readers should try to best match all experimental and environmental conditions, especially the flow rate, when choosing a removal rate to estimate setback distances. For a conservative approach, removal rates determined from area-sources under long-term effluent loading should be considered. Caution should be exercised when extrapolating the distances beyond the transport scales that the removal rates are derived from. Removal rates may reduce with distance, especially in fine grain aquifers and aquifers under long-term continuous input of effluent.

Table 16. Summary of the magnitude of removal rates for different subsurface media.

Category	Magnitude of removal rate λ log/m	Conditions
Soil	$>10^{1}$	Allophanic and pumice sand soils
	10^{0}	Most soil types
	10^{-1}	Clayey soil, clay loam and clayey silt loam
Vadose zone	10^{0}	Pumice sand, clay till, occasionally sand
	10^{-1}	Clay and silt, sand, sand-gravels, coarse gravels, fractured chalk and granite
Sand aquifers ($V < 2$ m/d)	10^{0}	Pumice sand aquifers
	10^{-2}–10^{-1}	Sand aquifers
Sand and gravel aquifers ($V < 3$ m/d)	10^{-1}	$x < 17$ m, clean and contaminated, occasionally 10^{-2} log/m and 10^{0} log/m
	10^{-2}–10^{-1}	$x < 177$ m (including river bank filtration)
	10^{-3}	$x = 183$–970 m, contaminated
	10^{-4}	$x = 210$–2930 m, contaminated
Sandy gravel aquifers ($V > 11$ m/d)	10^{-3}–10^{-2}	$x < 163$ m , clean
Coarse gravel aquifers ($V > 50$ m/d)	10^{-2}	Clean
	10^{-3}	Contaminated
Fractured rock aquifers	10^{-1}–10^{0}	Clean fractured clay till and fractured clayed shale saprolite
	10^{-2}–10^{-1}	Clean fractured gneiss
	10^{-2}	Contaminated sandstone
	10^{-3}–10^{-2}	Contaminated fissured chalk
Karst limestone aquifers	10^{-2}–10^{-1}	$x < 85$ m, contaminated
	10^{-3}	$x = 1250$ m, contaminated
	10^{-4}	$x = 5000$ m, contaminated

Acknowledgments

The author much appreciates the assistance provided by Miss Elizabeth Crone and Sarah Coxon, summer students from the University of Canterbury, for their help with literature search and data entry. Dr. Hilary Michie and Mr. Murray Close (Institute of Environmental Science and Research), and the three journal referees are thanked for their valuable comments on the manuscripts. Professors Mark Goltz, Markus Flury, Joseph Ryan, and Dr. Scott Bradford are acknowledged for their helpful discussions in some aspects of this study. This study was funded by the Foundation for Research, Science and Technology (New Zealand) under contract numbers CO3X0303 and ESRX0702.

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