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Setback distances between small biological wastewater treatment systems and drinking water wells against virus contamination in alluvial aquifers

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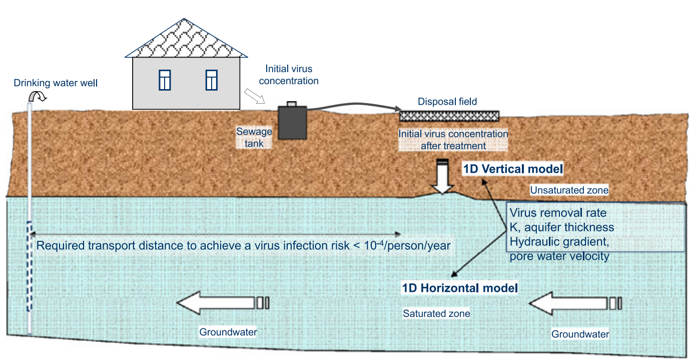
HIGHLIGHTS

- To ensure $<10^{-4}$ enteric virus infection/year/person, it needs a 12-log reduction.
- This would need a horizontal setback distance of 39–144 m in sand aquifers.
- It increases to 66–289 m in gravel aquifers and 1–2.5 km in coarse gravel aquifers.
- For unsuitably large setback distance, extra treatment is needed before disposal.
- Using on-site information, results help to guide decision making in rural planning.

GRAPHICAL ABSTRACT



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ABSTRACT

Contamination of groundwater by pathogenic viruses from small biological wastewater treatment system discharges in remote areas is a major concern. To protect drinking water wells against virus contamination, safe setback distances are required between wastewater disposal fields and water supply wells. In this study, setback distances are calculated for alluvial sand and gravel aquifers for different vadose zone and aquifer thicknesses and horizontal groundwater gradients. This study applies to individual households and small settlements (1–20 persons) in decentralized locations without access to receiving surface waters but with the legal obligation of biological wastewater treatment. The calculations are based on Monte Carlo simulations using an analytical model that couples vertical unsaturated and horizontal saturated flow with virus transport.

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Hydraulic conductivities and water retention curves were selected from reported distribution functions depending on the type of subsurface media. The enteric virus concentration in effluent discharge was calculated based on reported ranges of enteric virus concentration in faeces, virus infectivity, suspension factor, and virus reduction by mechanical-biological wastewater treatment. To meet the risk target of $<10^{-4}$ infections/person/year, a 12 $\log_{10}$ reduction was required, using a linear dose-response relationship for the total amount of enteric viruses, at very low exposure concentrations. The results of this study suggest that the horizontal setback distances vary widely ranging 39 to 144 m in sand aquifers, 66–289 m in gravel aquifers and 1–2.5 km in coarse gravel aquifers. It also varies for the same aquifers, depending on the thickness of the vadose zones and the groundwater gradient. For vulnerable fast-flow alluvial aquifers like coarse gravels, the calculated setback distances were too large to achieve practically. Therefore, for this category of aquifer, a high level of treatment is recommended before the effluent is discharged to the ground surface.

1. Introduction

Many waterborne disease outbreaks are caused by the consumption of groundwater that is contaminated by microbial pathogens (Beer et al., 2015; Beller et al., 1997; Borchardt et al., 2011; Craun et al., 2002; Fong et al., 2007; Jalava et al., 2014; Miettinen et al., 2001; Parshionikar et al., 2003). Faecal bacterial indicators are commonly used to indicate water contamination by pathogens even though pathogenic viruses and protozoa can have higher persistence than bacteria (Rose and Gerba, 1991). Protozoa have generally lower input concentrations and are one and two orders of magnitude larger in sizes than bacteria and viruses, respectively, so they are more likely to be filtered out (Farnleitner et al., 2010). Viral contamination tend to be overlooked due to the large volumes of water required for obtaining representative samples as well as the high costs associated with their analyses. However, recent studies have demonstrated that not only faecal bacteria- but also pathogenic viruses are widespread in groundwater, e.g., in the United States (Abbaszadegan et al., 2003; Borchardt et al., 2003; Borchardt et al., 2007; Borchardt et al., 2004; Fout et al., 2003). Virus-positive samples have even been found in the absence of bacteria (Borchardt et al., 2003; Frost et al., 2002; LeChevallier, 1996). In a survey of 448 groundwater sites in 35 US states, 31.5% sites were positive for at least one pathogenic virus type (Borchardt et al., 2003). Enteric viruses have also been detected in groundwater in many other developed countries (Gallay et al., 2006; Jung et al., 2011; Karamoko et al., 2006; Masciopinto et al., 2007; Powell et al., 2003) and developing countries (Guerrero-Latorre et al., 2011).

Leaching of pathogens from human and animal effluent and wastes through subsurface media is a major contributor to groundwater contamination. This has increased the need to establish safe setback distances between on-site disposal fields and drinking water supply sources (e.g., wells, springs, reservoirs), food-growing waters (e.g., shellfish and salmon farms), and recreational water bodies (e.g., lakes, bathing beaches). Setback distances, when properly determined, ensure the sustainable removal of pathogens by natural attenuation processes in subsurface media so that the quality of the receiving water is acceptable for specific purposes.

Subsurface media act as natural filters and buffers that can mitigate faecal contamination, but they vary widely in their ability to remove microbial contaminants. This is shown in the observed maximum horizontal travel distances of microbes. For example, injected bacteria traveled 14 km in a karst aquifer with a velocity of 250 m/h (Batsch et al., 1970), bacteria traveled 15 km at 167–190 m/d in chalk aquifer (Hutchinson, 1972), bacteriophages (phages) traveled 920 m in a contaminated coarse gravel aquifer (Noonan and McNabb, 1979), bacteria traveled 600 m in a contaminated sandy fine gravel aquifer (Harvey, 1991; Harvey and Garabedian, 1991), phages traveled 30 m in a contaminated coastal sand aquifer (Schijven et al., 1999), and phages traveled <6 m in a clean pumice sand aquifer (Wall et al., 2008).

In this paper, the term 'setback distance' is defined as the distance between a wastewater disposal field and a drinking water well in the direction of flow. Several examples in the United States (Azadpour-Keeley et al., 2003; Deborde et al., 1999), Australia (Geary and Pang, 2005), Canada (Dunn et al., 2014), and Italy (Masciopinto et al., 2007) show diverging management strategies for the choice of setback distances to protect down-gradient receiving waters. The scientific background for the design of setback distances is often unclear. Some states in the USA have adopted a setback distance of 30.5 m as a standard distance between wells and septic systems (Deborde et al., 1999). Likewise, many states in the U.S. recommend a vertical separation distance of 30–45 cm between the drain-field trench bottom and a limiting soil interface or groundwater (Karathanasis et al., 2006).

In 7 out of 10 Canadian provinces, a minimum of 4 $\log_{10}$ reduction of enteric viruses are required by law from the pollution source towards the point of water use, regardless of the concentration in the source water (Dunn et al., 2014). In many countries (e.g., Austria, Denmark, Germany Ghana, Indonesia, the Netherlands, UK), groundwater used for drinking is protected from other uses in the vicinity of the wells using a travel time of 50–60 days. Some faecal pathogens and in particular enteric viruses, however, were found to survive several months in groundwater. For example, Rotavirus can persist in groundwater up to seven months (Espinosa et al., 2008), and Adenovirus can remain infectious for at least one year in groundwater (Charles et al., 2009). Thus, resource management authorities and the public increasingly request more specific criteria for designing setback distances as they relate to different subsurface media.

Setback distances were previously estimated from different authors for some aquifer media (Table 2). Earlier estimates of setback distances were often based on reductions in microbial numbers from inactivation only (Yates and Yates, 1989), while later development considered total removal (attachment, straining, and inactivation) in the calculations (Charles and Ashbolt, 2004; Masciopinto et al., 2007; Masciopinto et al., 2008; Moore et al., 2010; Pang et al., 2005b; Pang et al., 2004; Schijven and Hassanizadeh, 2002; Schijven et al., 2006; van der Wielen et al., 2006; van der Wielen et al., 2008). Both unsaturated and saturated flow conditions were considered for estimating setback distances from septic tank systems, e.g., as part of the pre-development phase of the Groundwater Rule by the United States Environmental Protection Agency (Berger, 1994), (USEPA, 2006b), and other studies from different parts of the world (Gunnarsdottir et al., 2013; Kroiss et al., 2006; Moore et al., 2010). Usually, there are soils and vadose zones above the water table and depending on their thicknesses, the horizontal setback distances required can be significantly reduced (Charles and Ashbolt, 2004). Despite these past efforts, there is still a need for a more systematic evaluation of small wastewater treatment systems in remote areas for alluvial aquifers that depend on the vadose zone thickness and groundwater flow conditions (Charles and Ashbolt, 2004; Gunnarsdottir et al., 2013). In recent years, an extensive database of microbial removal rates for a wide range of soils, vadose zones, and aquifer media has become available. The database was established (Pang, 2009) based on an analysis of a large amount of field data published in the literature. With this available database, it is thus possible to determine groundwater setback distances in alluvial aquifers, considering the presence of vadose zones.

The objective of this study is to calculate setback distances in alluvial aquifers against contamination from pathogenic viruses, considering the thicknesses of vadose zone and aquifer and variable groundwater hydraulic gradient. Small biological wastewater treatment systems were considered for 1–20 persons in decentralized locations. As trenches are generally excavated below soils, we have excluded soils in our estimation of setback distances. Recommendations are also given for improving the performance of on-site treatment systems on virus removal for the case where the required setback distances cannot be met practically. A further objective of this study was hence to calculate the required total enteric virus $\log_{10}$ reductions for a given distance. A linear dose-response relationship was applied for the small disposal systems that was based on exposure estimates for total enteric viral numbers as expected from epidemiological data and concentrations in the faeces of infected people. These results can be used as a guideline to estimate the level of improvement needed for the performance of on-site treatment systems.

2. Material and methods

2.1. Water quality criteria for drinking water

The EU Drinking Water Directive (2015) requires drinking water to contain <1 *Escherichia coli* and *Enterococci* in any 100 mL sample, and the Drinking-water Standards for New Zealand requires drinking water to contain <1 *E. coli* in any 100 mL sample and <1 pathogenic protozoa/100 L (MOH, 2008). Also, immediate investigations must be undertaken if *E. coli* is found in any 100 mL sample of drinking water according to WHO Drinking water guidelines (WHO, 2011). These criteria, however, do not consider the infection risks associated with pathogenic viruses and other pathogenic microorganisms, which generally have low infectious doses and can be very persistent in water.

USEPA policy includes water quality criteria based on acceptable risks of infection which may vary, e.g., for different surface water treatment systems (USEPA, 2006a). The criterion to minimize the risk of infection below 10^{-4}/person/year was implemented in Dutch drinking water regulations (Bichai and Smeets, 2013), which was derived based on recommendations of the World Health Organization (WHO, 2011). This criterion was adopted in this paper. Health based targets are under ongoing discussion, as new data become available and clinical practice changes (Sinclair et al., 2015).

The water quality criterion used in this paper is described below. The dose-response relationship and the probability per case of viral infection was approximated by a linear relationship (WHO, 2004) using statistical exposure estimates of viral numbers derived from the total amount of available enteric viruses. Expected available enteric viruses in raw waste water of small disposal systems were derived from epidemiological wastewater data and observed viral concentrations in faecal samples of infected persons (see below for details). Based on the linear dose response relationship (Hurst, 2002), the expected infectivity from enteric viruses is given by

$$p_{inf} \approx pCW \tag{1}$$

where notations are given in Table 1. p in Eq. (1) was taken from the reciprocal of the minimum infectious dose of enteric viruses (Hurst, 2002). An extrapolation using non-linear models was found unsuitable due to sparse data availability (Haas and Eisenberg, 2001). Using the mean values plus twice the standard deviation of p and W (95-percentiles, Table 3C), a concentration in drinking water of $\leq 3.4 \times 10^{-7}$ total number of enteric virus particles/L was calculated by solving Eq. (1) for C. This concentration is required to fulfil the condition of $p_{inf} < 10^{-4}$ infections/person/year. In comparison, Regli et al. (1991) determined a Rotavirus concentration of 2.2×10^{-7} particles/L to fulfil the same condition. In contrast to the approach by Hurst (2002), the value of Regli et al. (1991) was determined using an extrapolation for the minimum infectious dose, which was 1.6 particles for Rotavirus. A value of 2 particles was used for the minimum infectious dose in this paper, which was taken from an overall estimate of enteric pathogenic viruses (Hurst, 2002). Furthermore, a consumption of 2 L/person/day was assumed by Regli et al. (1991), which was twice as much as assumed in this paper. To design a target level of pathogenic virus reduction, one needs to know enteric virus concentrations at the source of the contaminants. This is described next.

Table 1
Notations.

Symbol	Parameter	Unit
p_{inf}	Probability per case of enteric viral infection based on dose-response relationship	–
p	Probability of infectious enteric virus particles	–
C	Enteric virus concentration in drinking water well	Particle/L
C_0	Input enteric virus concentration of biologically treated wastewater	Particle/L
S_{fae}	Enteric virus concentration in human faeces	Particle/m^3
i	Virus infectivity, i.e., fraction of infectious enteric viruses over total enteric virus population	–
s	Suspended faeces in sewage effluent	m^3/L
r	Virus reduction, i.e., fraction of virus concentration in raw over treated wastewater	–
r_a	Required enteric virus reduction to achieve the infection risk target	
α	Infection risk target	$<10^{-4}$ infections/person/year
W	Daily volume of water consumption per person	L/d/person
Q_{in}	Infiltration rate of treated effluent into vadose zone	L/d
q_{in}	Infiltration rate per area	L/m^2/d
q	Volumetric flux density of water	m/d
θ_e	Effective porosity	–
θ	Volumetric water content	–
K	Hydraulic conductivity	m/d
N	Van Genuchten model parameter	1/m
θ_r	Residual water content	–
θ_s	Saturated water content	–
Ψ	Water pressure potential	m
α_l	Longitudinal dispersivity	m
λ_s	First-order virus removal rate	ln/m
v	Pore-water velocity	m/d
μ	Virus inactivation rate	ln/d
x	Distance in the direction of groundwater flow	m
z	Depth below ground surface	m

2.2. Virus concentrations in small biological wastewater treatment systems

Raw wastewater can contain significant numbers of infectious agents, and microbial reduction by small biological wastewater treatment plants is limited. The treatment of such wastewater treatment plants is still considered an essential first barrier for the reduction of microbial pathogens (Table 4) and is required before undertaking further treatment steps. A wastewater treatment system also helps to avoid the direct contact of animals with raw effluent, which may lead to the spread of infectious diseases over faunal vectors, e.g., rats, mice, birds, insects (Mathys, 1998).

Little information is available on the actual measured enteric virus concentrations in small biological wastewater treatment systems in decentralized locations (Farnleitner et al., 2010; Canter and Knox, 1985), yet more data is available for centralized systems (Dahling et al., 1989; Greening et al., 2000; Lodder and Husman, 2005). Enteric virus concentrations in small biological wastewater treatment systems in decentralized locations are expected to vary widely compared with homogenised effluent in centralized treatment systems, because the concentrations will depend on whether there are infected people in the dwellings. In general, enteric viruses will occur less frequently in smaller systems than in larger communal wastewater systems. However, the peak concentrations of enteric viruses in small biological wastewater treatment systems can be much higher than those in centralized sewage systems because of less dilution with non-contaminated parts of the wastewater (Farnleitner et al., 2007). Moreover, the rates of excretion and duration of infection can vary (Charles et al., 2003). Large numbers of enteric viruses are excreted in the faeces of infected people often for prolonged periods of time (2–3 months), and larger numbers are excreted by younger children (Gerba, 2000). Even those who are not clinically ill may excrete significant numbers of pathogens (Gerba, 2000).

For simulating the setback distances from small biological wastewater treatment systems designed for 1–20 persons, the enteric virus concentrations in the treated effluent were calculated by

$$C_0 = (S_{fae} \cdot \mathrm{i} \cdot \mathrm{s})/(\mathrm{r}) \tag{2}$$

Refer to Table 1 for notations and Table 3A for input variables. The mean and standard deviation of C_0 were calculated using reported input values (Table 3A). It was assumed that all persons are infected at the same time. Using Eq. (1) and the mean values listed in Table 3C, the required mean reduction in enteric virus concentration to achieve the infection risk target (α) of $<10^{-4}$ infections/person/year were thus calculated by

$$r_a \geq \frac{\alpha}{pC_0W} \tag{3}$$

Notations are listed in Table 1. The required mean enteric virus reduction to achieve the water quality criterion of $\leq 3.4 \times 10^{-7}$ enteric virus particles/L is 12 $\log_{10}$. This value was considered a good realistic estimate (Haas et al., 1999) in contrast to the 95th percentile value, which is based on respective worst conditions and is therefore very conservative. A required mean virus reduction of 12 $\log_{10}$ was therefore considered for the simulations of setback distances. For more details see also Kroiss et al. (2006).

2.3. Water flow and virus transport model for simulating setback distances

A 1-D water flow model was used to simulate vertical unsaturated flow coupled with horizontal saturated groundwater flow. The rate at which water moves in one dimension through the unsaturated zone was simulated according to (Nielsen et al., 1986),

$$q = K(\theta) \cdot \frac{\partial \Psi}{\partial z} \tag{4}$$

Assuming a steady state flow condition, the change of water pressure over depth below the ground surface ($\frac{\partial \Psi}{\partial z}$) was set to 1. The pore water velocity in the vadose zonewas calculated next as

$$v = q/\theta \tag{5}$$

$K(\theta)$ was calculated by using the van Genuchten (1980) model with the parameters N, θ_r, and θ_s, (Nielsen et al., 1986), refer to Table 1 for notations. Eqs. (4) and (5) were solved with the condition that q is equal to the rate at which effluent water infiltrates over the area for infiltration (q_{in}). For gravel and coarse gravel media, a, N, θ_r, and θ_s were derived from fitting the van Genuchten model to measured water retention curves in coarse gravel media (Fig. 1). For simulating horizontal flow, the Darcy equation for the saturated zone was solved as a function of

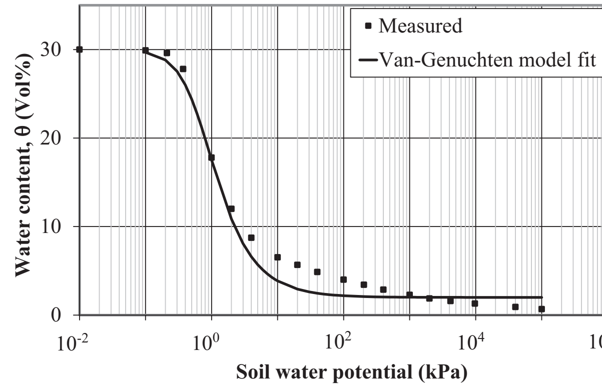

Fig. 1. Water retention curves used for simulating virus transport in unsaturated coarse gravel (data provided by the Federal Office for Water Management, Petzenkirchen, Austria).

K, of the groundwater gradient, and of the effective porosity θe, which was calculated from subtracting θ_s and θ_r (Kinzelbach, 1987; Bear, 1988). The virus transport in both the unsaturated and saturated zones was calculated by the 1-D advection-dispersion equation (Kinzelbach, 1987; van Genuchten, 1981), and couples with first-order virus removal and inactivation rate:

$$log_{10}\frac{C}{C_0} = \frac{x}{2.3}\left(1 - \sqrt{1 + 4\alpha_l\left(\frac{\lambda_s \cdot v + \mu}{v}\right)}\right)/2\alpha_l \tag{6}$$

Notations are listed in Table 1. This equation is for steady-state groundwater flow and virus transport conditions and was solved in the form of $\log_{10}$ reduction of C relative to C_0 at a certain distance in the direction of groundwater flow. The initial enteric virus concentration (C_0 in Eq. (6)) was taken from the enteric virus concentration in the effluent water calculated in Eq. (2). The enteric virus concentration after vertical infiltration and transport towards the bottom of the vadose zone was simulated using Eq. (6). This concentration was then divided by the aquifer thickness multiplied by the width of the infiltration area and was used as the new initial enteric virus concentration for calculating the horizontal enteric virus transport in saturated groundwater using Eq. (6).

The most conservative mean values of λ_s were selected for a virus indicator from the databases of Pang (2009) for sand, gravel, and coarse gravel vadose zone and aquifer media (Table 3B), and setback distances as a function of vadose zone thickness were simulated. A certain extent of virus inactivation occurred naturally during the field experiments reported by Pang (2009) and was lumped into λ_s. In addition, virus inactivation was considered in the simulations by setting μ according to ranges found in the literature at 10 °C (Eq. (6), Tables 3B and 6). The analytical water flow and virus transport equations were solved using MATLAB and Statistics Toolbox Release 2015b (The MathWorks, Inc., Natick, Massachusetts, United States).

2.4. Simulations of setback distances

The required setback distances from small biological wastewater treatment systems (1–20 persons) to achieve safe drinking water were simulated for different sets of hydrological input variables. Values of hydraulic conductivities and van Genuchten parameters for the vertical flow in the unsaturated zone were selected for sand, gravel, and coarse gravel media (Table 3B). Vadose zone thickness values of 1, 3, 5, 10, and 20 m as well as aquifer thickness values of 3, 5, and 10 m and groundwater gradients of 0.001, 0.005, 0.01 and 0.05 were used. The combinations of the different hydrological input variables resulted in a total of 144 simulation cases. The equations for the vertical unsaturated and horizontal saturated flow and virus transport (Eqs. (5) and (6)) were solved by drawing random input variables from distribution functions, as specified in Table 3. The simulations were repeated 4000 times for each case following the Monte Carlo framework. This value was chosen because further iterations showed no significantly different results. The setback distances were then determined from the 95th percentiles of the simulated distances in the direction of flow. In addition, the $\log_{10}$ reductions of virus concentrations (95th percentiles) were simulated with the vadose zone thickness of 1 m and 20 m and the groundwater gradients of 0.01 and 0.001, respectively.

3. Results and discussion

3.1. Simulated setback distances

Fig. 2 and Table 7 display the simulated 95th percentile setback distances for achieving 12 $\log_{10}$ virus reductions in sand, gravel, and coarse gravel aquifers for vadose zone thickness of 1–20 m and groundwater gradient of 0.001–0.05. Simulated setback distances range 39–144 m in sand aquifers, 66–289 m in gravel aquifers, and 1–2.5 km in coarse gravel aquifers. The setback distances in sand aquifers predicted without the use of colloid filtration theory are in agreement with the predictions of van der Wielen et al. (2008) (110 m, Table 2), but not in agreement with Schijven et al. (2006) (206–418 m, Table 2) and van der Wielen et al. (2006) (276 m in the anoxic aquifer). Both Dutch authors applied colloid filtration theory for the well sorted, uniform dune sands. As stated by van der Wielen et al. (2008), the reason for the discrepancy between these studies is that extremely low inactivation rates and collision efficiencies were assumed by Schijven et al. (2006) and van der Wielen et al. (2006).

On average, when groundwater gradient increases from 0.001 to 0.05, the simulated setback distance extends by a factor of 1.5 in sand aquifer or coarse gravel aquifer and by a factor of 2.5 in gravel aquifer (Table 7). On average, when the thickness of the vadose zone decreases from 20 to 1 m, the simulated setback distance extends by a factor of 1.5, 1.3, and 1.0 in sand aquifers, gravel aquifers, and coarse gravel aquifers, respectively. Varying the saturated aquifer thickness from 3 to 10 m hardly affected the simulated setback distances. Thus results are only shown for an aquifer thickness of 3 m in Table 7, Figs. 2 and 3. The simulations suggested that setback distances of 1 km and more were required for a 12 $\log_{10}$ virus reduction in coarse gravel aquifers, even when the vadose zone thickness was set to 20 m. Setback distances of 1 km and more were also found for achieving a 7 $\log_{10}$ virus reduction in a limestone aquifer (Abbaszadegan et al., 2003). Much larger setback distances were reported by others, i.e., up to 3.8 km in a contaminated coarse gravel aquifer (Pang et al., 2005a), 3–8 km in fractured limestone aquifers (Masciopinto et al., 2007; Masciopinto et al., 2008). Fast-flow aquifers like coarse gravels, fractured rocks, and karst limestones are vulnerable for microbial contamination and require very large setback distances. For these types of aquifer media, a high level of treatment is recommended before the effluent is discharged to the ground surface.

3.2. Virus reduction by vadose zone and aquifer passage

In many cases, the required setback distances may not be feasible practically or economically; thus, additional treatment is needed. The required virus $\log_{10}$ reduction that must be achieved by additional measures was therefore simulated for a given distance of 20–500 m and groundwater gradients of 0.001 and 0.01, respectively (Fig. 3). Model results suggest that virus reduction is at least 12 $\log_{10}$ in sand aquifers

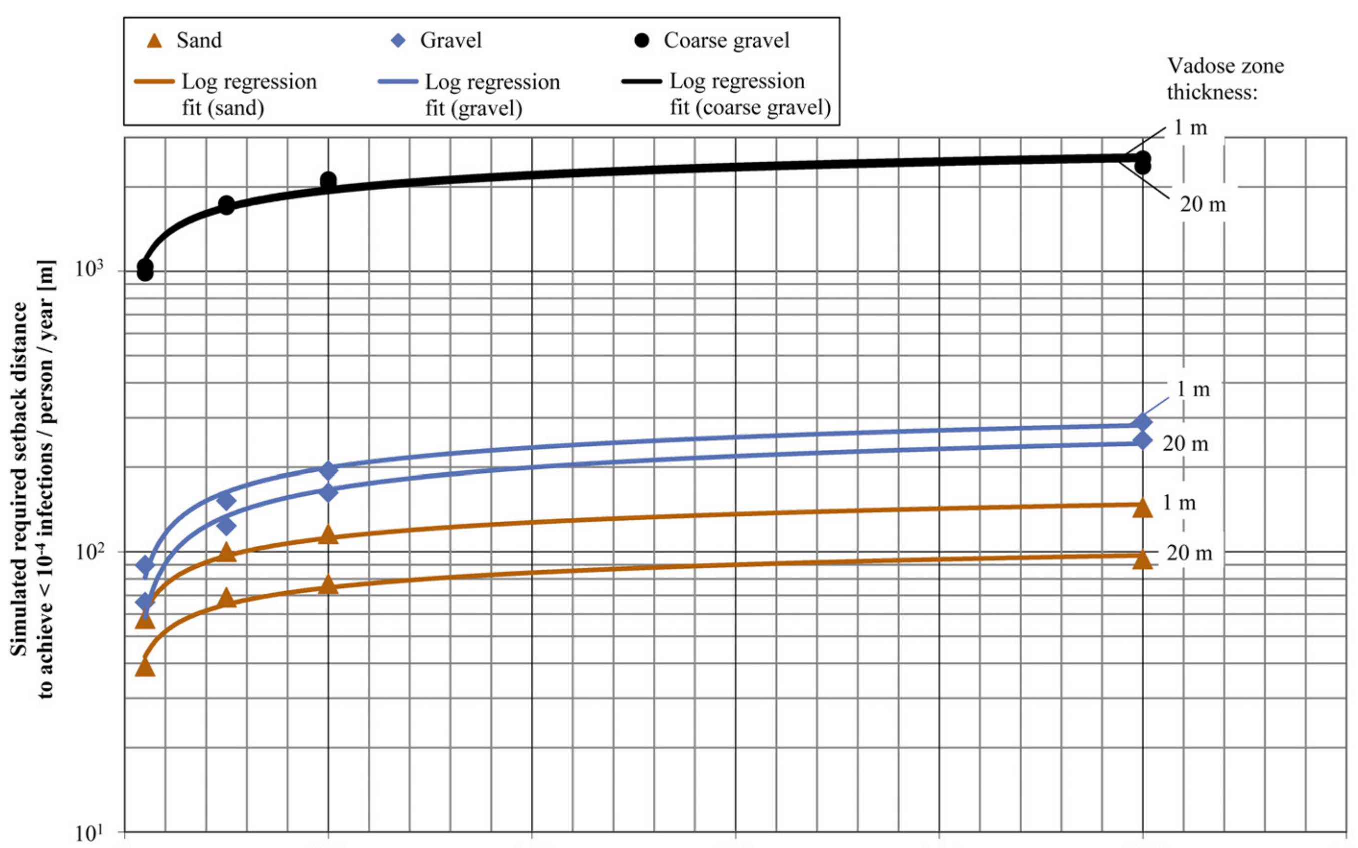

Fig. 2. Simulated required setback distances (95th percentiles) for achieving a 12 $\log_{10}$ virus reduction in sand, gravel, and coarse gravel assuming the same lithology in both vadose zone and aquifer. Input parameters are given in Table 1.

Table 2
Review of previously reported setback distances in groundwater.

Reference	Aquifer media and study area	Reduction in concentration	Criteria	Reduction mechanisms	Method	Setback distance (m)
Yates and Yates (1989)	Tucson Basin, unspecific aquifer	7 $\log_{10}$ reduction in viruses		Inactivation	Modeling	15–300
Berger (1994); Berger (1994)	Sandy loam, groundwater 10–15 °C	11 $\log_{10}$ reduction in viruses	$<2 \times 10^{-7}$ virus/L so that virus infection $<10^{-4}$/p/y	Inactivation	Modeling	160–325
Pang et al. (2004)	Uncontaminated pumice sand aquifer, Rotorua, New Zealand	10 $\log_{10}$ reduction in viruses for drinking water 5 $\log_{10}$ reduction in *E. coli* for recreation water	<1 virus/100 L in drinking water <126 *E. coli*/100 mL for recreation water	Total removal	Modeling	48 from well 16 from bathing beach
Gunnarsdottir et al. (2013)	Coarse aquifer media at 5 °C	9 $\log_{10}$ reduction in Noroviruses	$<1.8 \times 10^{-7}$ virus/L so that virus infection $<10^{-4}$/p/y	Total removal	Modeling	900
Pang et al. (2005a); Pang et al. (2005b)	Sand and gravel aquifers	7 $\log_{10}$ reduction in viruses and faecal bacteria	zero virus/100 L, zero faecal bacteria/100 mL	Total removal	Experimental	33–3889
Schijven and Hassanizadeh (2002); Schijven et al. (2006)	Sand aquifer, the Netherlands	9 $\log_{10}$ reduction in viruses	$<1.8 \times 10^{-7}$ virus/L so that virus infection $<10^{-4}$/p/y	Total removal	Modeling	153–357 206–418
van der Wielen et al. (2006)	Oxic and anoxic sand aquifers, the Netherlands		virus infection $<10^{-4}$/p/y	Total removal	Modeling	54–84 oxic aquifer 276 anoxic aquifer
van der Wielen et al. (2008)	Anoxic coarse sand aquifer, the Netherlands	8.8 $\log_{10}$ reduction of Enterovirus and 9.3 $\log_{10}$ reduction of Reovirus	$<1.2 \times 10^{-6}$ virus/L so that virus infection $<10^{-4}$/p/y	Total removal	Modeling	110
Abbaszadegan et al. (2003)	Limestone aquifer, USA	Samples that were tested positive with cell culture and RT-PCR were analysed for the distance to a source of contamination			Experimental	1000
Masciopinto et al. (2007)	Fractured limestone aquifer, Italy				experimental and modeling	3000
Masciopinto et al. (2008)	Fractured limestone aquifer, Italy	7 $\log_{10}$ reduction in viruses	Simulated lowest removal rate 0.1 ± 0.06 d^{-1}, groundwater velocity V = 50 m/d	Total removal	Modeling	8000 ± 4800
Kvitsand et al. (2015)	Norwegian riverbank field site	8.7 $\log_{10}$ reduction in viruses	$<1.8 \times 10^{-7}$ virus/L so that virus infection $<10^{-4}$/p/y	Dilution, dispersion, irreversible attachment	Modeling	174

over 100 m and in gravel aquifers over 200 m, but it is only 1.6 $\log_{10}$ in coarse gravel aquifers over 200 m. The vadose zone was shown to be effective in reducing virus concentrations, in particular, for sandy media. Modeling results suggest that, when the thickness of the vadose zone decreases from 20 to 1 m, the virus reduction in vadose zone decreases by 4 $\log_{10}$ in sand, 1–3 $\log_{10}$ in gravel media, and only 0.4 $\log_{10}$ in coarse gravel media (Fig. 3). Possible actions to achieve additional virus reduction include improving the level of treatment by additional disinfection steps, e.g., UV treatment.

3.3. Model assumptions and scope of applications

In the flow and virus transport simulations of setback distances and virus $\log_{10}$ reductions, some assumptions were made, which will ultimately have implications on the applications of the results. The results of this paper are only applicable to alluvial aquifers and riverbank filtration sites in moderate climate regions. The setback distances estimated are conservative as the lowest virus removal rates (λ_s) given in the database of Pang (2009) were used in the model simulations. The properties of the aquifer materials and the pore water velocity play a critical role in influencing virus transport but these effects have already been encompassed in the removal rate itself. This is because removal rates that were used in this paper were derived from field studies (Pang, 2009). As for the virus inactivation rates (μ), they depend on many influencing factors, such as chemical and physical conditions and the microbial heterotrophic activity (Hurst, 1991). The temperature has the largest impact; for example, a higher inactivation rate was found with increasing temperature for MS2 phages (Gerba et al., 1991). In moderate climates, where the groundwater temperature is around 10 °C, the impact is expected to be low. The impact on μ at higher temperatures was therefore neglected in the simulations.

The focus of this paper was on the wastewater discharge of small households (1–20 persons). As in this case the wastewater discharge will be relatively small compared to the volume of the groundwater aquifer, the change in groundwater gradient due to the infiltration of wastewater was assumed to be negligible. It was further assumed that the subsurface medium was homogeneous and isotropic, the infiltration rates and concentrations at the inlet were constant over time, and virus particles at the point of infiltration were mixed over the full aquifer thickness. At real field sites, however, subsurface media are typically heterogeneous. The simulations of the setback distances were conducted assuming a saturated thickness of the aquifer of 3–10 m. A greater thickness was not considered, as the discharged sewage water is commonly transported in the upper part of the saturated aquifer, and vertical mixing over the entire saturated thickness of the aquifer only takes place at very large transport scales. As the wastewater discharge of small households most likely is a continuous process, the assumption of complete vertical mixing over 3–10 m thickness of the aquifer is reasonable.

In order to compensate for the uncertainties arising from these simplified assumptions, the random nature of the model variables was accounted for by using a Monte Carlo framework. In addition, the discharge rate and enteric virus concentration in effluents are usually highly variable over time. Due to the uncertainty of viral source concentrations and due to limited data sets, a linear dose-response relationship based on the approach by Hurst (2002) was applied. This

Table 3
Input variables and ranges for simulating virus concentrations in effluent of small biological wastewater treatment systems and in groundwater. For notations see Table 1.

A. Input variables associated with wastewater treatment and septic tank system

Parameter	Units	Mean	Standard deviation	Statistical distribution	Reference
S_{fae}	N/m^3 faeces	3.4×10^{17}	2.4×10^{17}	Triangular	From min-max and peak values in Table 5
i	Fraction of total virus population	5.0×10^{-1}	2.0×10^{-1}	Triangular	Gantzer et al. (1998) for Enterovirus
s	m^3 faeces/L	1.0×10^{-6}*	0*	Constant	
r	–	75	44	Triangular	From min-max and peak values Table 4
C_0	N/L	2.3×10^{9}	1.0×10^{10}	Triangular	Eq. (4)
Q_{in} per 4-person-household	L/d	6.0×10^{2}*	0	Constant	
q_{in}	L/m^2/d	3.0×10^{1}		Constant	

B. Input variables for groundwater flow and virus transport simulations

Parameter	Units	Subsurface media	Minimum	Maximum	Statistical distribution	Reference
λ_s in vadose zones	ln/m	Sand	4.0×10^{-1}	2.5×10^{0}	Uniform	
		Gravel, coarse gravel	3.0×10^{-1}	1.2×10^{0}		
λ_s in saturated zones	ln/m	Sand	4.0×10^{-1}	5.0×10^{-1}		
		Gravel	4.0×10^{-3}	2.0×10^{0}		
		Coarse gravel	3.0×10^{-3}	1.0×10^{-1}		
		Riverbank sand and gravel	4.0×10^{-2}	4.0×10^{-1}		
μ	ln/d	Sand, gravel, coarse gravel	5.8×10^{-3}	2.4×10^{-1}	Uniform	Table 6
Vadose zone thickness	m	Sand, gravel, coarse gravel	1.0×10^{0}	1.0×10^{1}	–	
Aquifer thickness	m		3.0×10^{0}	1.0×10^{1}	–	
Groundwater gradient	–		1.0×10^{-3}	5.0×10^{-2}	–	

Parameter	Units	Subsurface media	Mean	Standard deviation	Statistical distribution	Reference
α_l in vadose zones	m		5.0×10^{-2}	0	Log-normal	
α_l in saturated zones	m		9.8×10^{-1}	8.9×10^{-1}		Gelhar et al. (1992)
K in saturated zones	m/d	Sand	7.1×10^{0}	3.7×10^{0}		Carsel and Parrish (1988)
		Gravel	3.0×10^{1}	1.7×10^{1}		Burger and Belitz (1997)
		Coarse gravel	1.5×10^{3}	1.3×10^{3}		Jussel et al. (1994)
θ_s	–	Sand	4.0×10^{-1}	6.0×10^{-2}	Uniform	Carsel and Parrish (1988)
		Gravel and coarse gravel	3.0×10^{-1}			Fig. 1
θ_r	–	Sand	4.5×10^{-2}	1.0×10^{-2}		Carsel and Parrish (1988)
		Gravel and coarse gravel+	2.0×10^{-2}	0	Uniform	Fig. 1
N	1/m	Sand	2.7×10^{0}	3.0×10^{-1}		Carsel and Parrish (1988)
		Gravel and coarse gravel+	2.0×10^{0}	0	Uniform	Fig. 1

C. Input variables for QMRA

Parameter	Units	Mean	Standard deviation	Statistical distribution	Reference
p	Infections per enteric virus particle	5.2×10^{-1}**	1.9×10^{-1}**	Triangular	Hurst (2002); Hurst et al. (1996)
W	L/person/d	5.0×10^{-1}	2.0×10^{-1}	Triangular	Mons et al. (2007)

* Based on a mean faeces production of 150 g and a mean wastewater production of 150 L/person/day and a faecal density of 1 g/cm^3.
** Defined as the reciprocal of the minimum infectious virus doses (WHO, 2004c).
+ Constant values of van Genuchten parameters were assumed in gravel and coarse gravel media because a sensitivity analyses showed that they had little effect on the simulated setback distances (not shown).

approach does not discriminate between varying infectivity of different viral lineages but assumes a linear and constant dose-response relationship for the total number of all infectious enteric viral particles encountered. The use of a linear dose response relationship is especially recommended by the WHO drinking water guidelines (WHO, 2004) for low exposure situations in case of limited data on exposition. More recent studies found that the probability of infection after ingesting one organisms is more than an order of magnitude greater than previously recognized (Messner and Berger, 2016; Messner et al., 2014). These drinking water infection risk estimations, however, used high dose human subject data to extrapolate to low dose drinking water exposure. The approach by Hurst

Table 4
Log_{10} reductions of virus and bacteriophages in wastewater treatment systems.

Virus type	Type of wastewater treatment	Characteristics	Log_{10} reduction	Reference
Virus undifferentiated	Mech. & biological	–	1–2	Grabow (1968)
Virus undifferentiated	Mech. & biological	–	>1	Berg (1973)
Enterovirus	Mech. & biological	–	0.8	Rolland et al. (1983a); Rolland et al. (1983b)
Enterovirus	Mech. & biological	–	1.7–2.1	Antoniadis et al. (1982), Payment et al. (1986)
Enterovirus	Mech. & biological	–	1.3	Lewis and Metcalf (1988),Leong (1983)
Enterovirus	Mech. & biological	–	1.2	Irving and Smith (1981)
Enterovirus	Mech. & biological	–	0.6–2.0	USEPA (1992b)
Poliovirus	Mech. & biological	–	1.0–1.3	Robeck et al. (1962)
Adenovirus	Mech. & biological	–	0.8	Irving and Smith (1981)
Coxsackie	Mech. & biological	–	4.7	Carlson (1967)
Poliovirus	Mech. & biological	–	1.0	Carlson (1967)
Enterovirus, coliphages	Constructed wetland	Aqua culture system, 4 day retention time	1.7	Karpiscak et al. (1996)
Enterovirus	Constructed wetland	Rhizosphere conditioning	0.7–1.0	Lopez and Warnecke (1988)
Poliovirus	Constructed wetland	Rhizosphere conditioning, 12–30 °C	2.0–3.0	Gersberg et al. (1987)

Table 5
Reported virus abundance in human faeces; derived values in brackets.

Virus species	Virus type	Known number of serotypes	Log_{10}/g faeces	Log_{10}/L raw and treated effluent[b]	Reference
Enterovirus[a]	Enterovirus	>67	7 (6–8)		Rotbart (1995)
			5–12		Gerba (2000)
				≤3.5 (raw)	Sedmak et al. (2003)
	Poliovirus	3	3–6.5		Melnick and Rennick (1980), Rotbart (1995)
	Coxsackie A virus	22	2–5.5		
	Echovirus	34	2–5.5		
Hepatovirus[a]	Hepatitis A virus	1	(>6)[f], up to 10		Cederna and Stapleton (1995); Walter (2000)
Caliciviridae[c]	Norwalk virus	1	(Up to 12)		Petric (1995); Walter (2000)
	NLV and SRSV [c]	~50	(7; 6–10[g])	4; 3–7[g] (raw)	Petric (1995)
Astroviridae[d]	Human Astrovirus	8	(>6[f], 8)		Petric (1995); Walter (2000)
Reoviridae[e]	Human Rotavirus	A–C	(up to 11, 12, 6–9[h])		Christensen (1995); Walter (2000)
			5–12		Gerba (2000)
			Up to 12	Mean 6 Max 10.9 (treated)	Charles et al. (2003)
Enteric Adenovirus	Enteric Adenovirus types 40 and 41	2	(11)		Leclerc et al. (2004)
	Enteric Adenovirus		5–12		Gerba (2000)

[a] Belonging to the Picornaviridae (single-stranded RNA+).
[b] Referring to septic-tank systems if not stated otherwise.
[c] Norwalk like viruses and small round structured viruses include virus types such as Sapporo (classical "Calicivirus"), Hawai, Snow Mountain, Taunton, Osaka Virus. Many SRSVs are summarized as gastroenteric viruses or as *Picorna-parvo like agents*.
[d] Single-stranded RNA+, from the replication strategy familiar to the Picorna virus type.
[e] Double-stranded spanned RNA.
[f] Reported detection limit of the electronic microscope.
[g] Estimated from concentrations in raw wastewater (Medema et al., 2003) assuming a prevalence of 0,1% and 0.2 g faeces/200 L.
[h] As [g] but with assumed prevalence from 0.01%–1%.

(2002) was thus decided to be acceptable because of the absence of low dose data. This approach allowed deriving statistically sound estimates for the expected numbers of enteric viral exposition concentrations in raw sewage of small systems from epidemiological data and enteric viral concentrations in faeces.

Table 6
Virus inactivation rates per day in groundwater and wastewater at 10 °C.

Habitat	Intestinal virus type			Bacteriophage			Reference
	Polio 1	Echo 1	HAV	PRD 1	MS2	FRNA	
Groundwater	0.18[a]	0.24[a]			0.16[a]		Yates et al. (1985)
Wastewater	0.03[b]			0.054[b]	0.077[b]		Blanc and Nasser (1996)
	0.11[c]		0.17[c]	0.051[c]	0.091[c]		
Groundwater					0.063[d]		Yates et al. (1985)
				0.01[e]	0.0058[e]		Yahya et al. (1993)
				0.1[f]	0.1[f]		
	0.11		0.19	0.025	0.11		Blanc and Nasser (1996)
	0		0.10			0	Nasser et al. (1993)
Wastewater	0.046[g]		0.17[g]			0[g]	
	0.0077[h]		0.12[h]			0.031[h]	
Groundwater	0.01[i]						Matthess et al. (1988)
	0.032[j]						
	0.013						

[a] At 12 °C.
[b] Wastewater after preliminary and biological treatment.
[c] Wastewater after preliminary, biological and further treatment.
[d] At 4 °C.
[e] At 7 °C.
[f] At 7 °C.
[g] Wastewater after preliminary treatment and sterilization.
[h] Wastewater after preliminary treatment.
[i] Sterilized groundwater.
[j] Deionized groundwater.

Note that the chosen linear dose-response approach is conservative by its very nature, and the derived 12 log_{10} required reduction for the total number of enteric viruses thus has to be considered a robust reduction target for this kind of wastewater disposal. In comparison, according to the dose-response relation of Rotaviruses (Regli et al., 1991) and according to (Berger, 1994) and (USEPA, 1992a), where the comment was made that a typical septic tank effluent could contain 10^{-4} pathogenic viruses/L, a 11 log_{10} reduction would be required. This level of reduction is 4 orders of magnitude greater than the 7 log_{10} reduction used in previous studies (Masciopinto et al., 2008; Pang et al., 2005a; Pang et al., 2004; Yates and Yates, 1989). In fact, the required enteric virus reduction varies over time and with population densities.

As shown by Zessner et al. (2007), alluvial aquifers dominated by sand and gravel are mainly vulnerable against virus contamination if drinking water quality is considered. The reasons are that in general high flow velocities and reduced filtration and adsorption capacities are present in such aquifers. In case of porous media with smaller

Table 7
Simulated 95th percentile setback distances from a small biological wastewater treatment system (1–20 persons) required for a 12 log_{10} viral reduction. See Table 3 for the input parameters; the aquifer thickness was set to 3 m.

Vadose zone thickness	Groundwater gradient	Setback distance		
[m]	[−]	[m]		
		Sand	Gravel	Coarse gravel
1	0.001	58	90	1039
	0.005	100	152	1744
	0.010	116	194	2064
	0.050	144	289	2521
10	0.001	50	76	1030
	0.005	84	125	1786
	0.010	99	184	2105
	0.050	119	259	2496
20	0.001	39	66	984
	0.005	69	124	1699
	0.010	77	163	2121
	0.050	94	249	2367

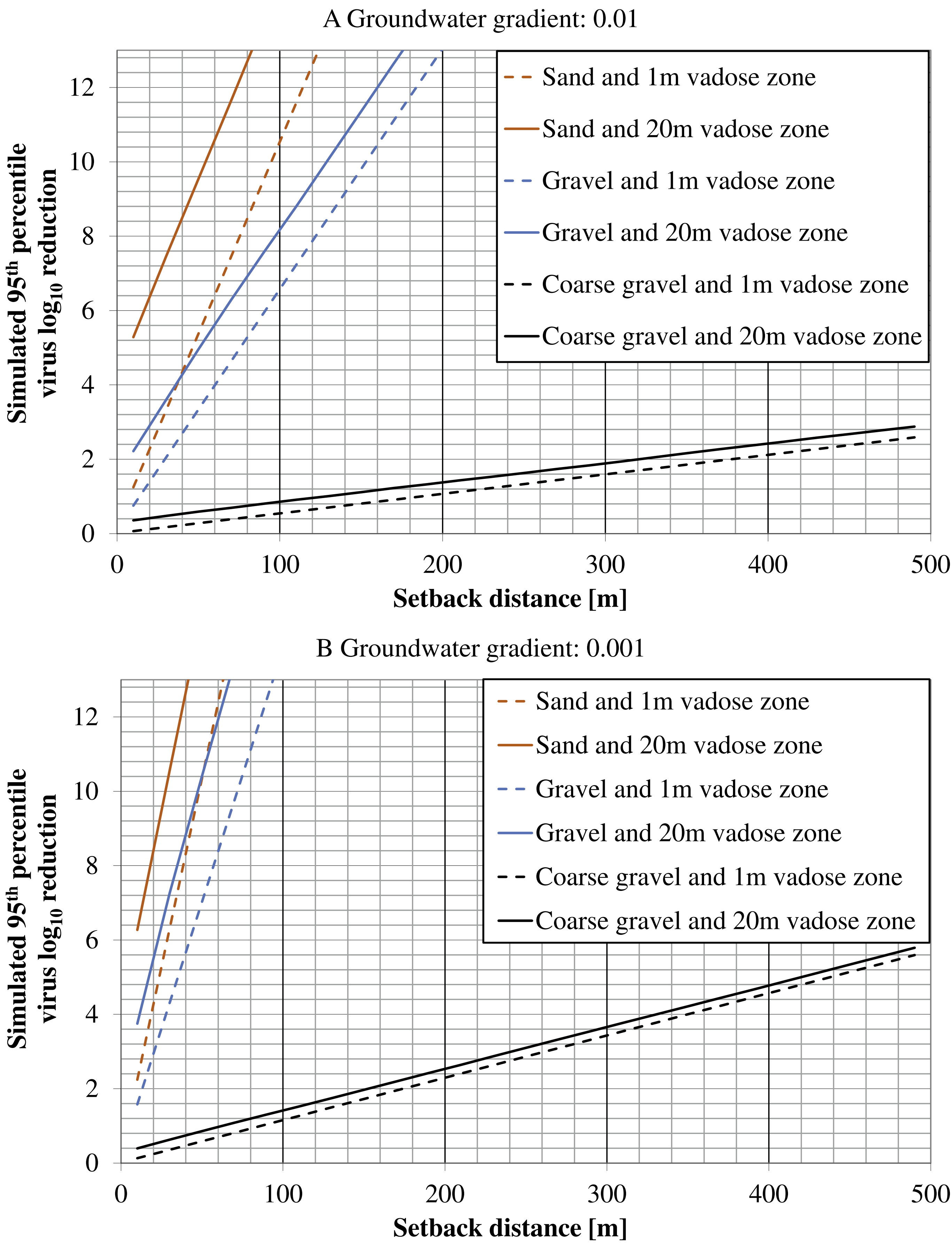

Fig. 3. Simulated 95th percentile virus $\log_{10}$ reduction by passage in vadose zone and aquifer of sand, gravel, and coarse gravel as functions of setback distance for a vadose zone thickness of 1 m and 20 m; (A) groundwater gradient 0.01, (B) groundwater gradient 0.001. Aquifer thickness was set to 3 m. Input parameters are listed in Table 1.

grain sizes, dilution of wastewater is low, and thus chemical pollution and oxygen depletion can cause negative impacts on ground water quality. For loamy sand aquifers and sandy loam aquifers, for instance, setback distances should therefore also be determined based on these parameters.

3.4. Recommendations for checking the feasibility and required setback distance at a site

Specific information are required in order to decide if treated wastewater can be discharged to the ground. These include the location of groundwater protection and restoration areas, the location of water supply sites, geological maps, the amount and type of wastewater, evidence that treated wastewater can be discharged to the ground, the initial level of groundwater pollution, and, the oxygen content of groundwater. In the following situations, it is not recommend to discharge treated wastewater to the ground: (1) there is a public canalisation system required by law; (2) the wastewater discharge into a watercourse is technically, economically and hygienically justifiable and there is no clear economic advantage of discharging the wastewater to the ground; (3) the site is located near a drinking water protection zone or a groundwater restoration area; (4) the vadose zone thickness is <1 m; and (5) in case of closed settlements without centralized water supply.

In order to estimate the required setback distances from the results of this paper, information about the vadose zone thickness, the saturated aquifer thickness, the groundwater gradient and the texture of the aquifer media are required. As the spatial distribution of the texture class can be heterogeneous, the predominant class can be used as input variable for determining the required setback distance based on the results of this paper. Therefore it is recommended to determine the texture class at several locations within the study area.

4. Conclusions

In this paper, a systematic health risk target-based modeling approach is presented for calculating setback distances from wastewater disposal fields to the points of drinking water use of alluvial aquifers. The results apply for small biological wastewater treatment systems in decentralized locations without access to centralized sewer systems. The simulated horizontal setback distances required for achieving 12-log reduction of the total numbers of enteric viruses vary widely, ranging 39–144 m in sand aquifers, 66–289 m in gravel aquifers and 1–2.5 km in coarse gravel aquifers. It also varies for the same media, depending on the thickness of the vadose zone and the groundwater gradient. The aquifer type was shown to have the largest impact on the simulated setback distances, which are 17–28 times larger in coarse gravel aquifers than in sand aquifers. The groundwater gradients were varied from 0.001 to 0.05 in the simulations, resulting in a 2.5 times larger setback distance at the highest gradient than at the lowest gradient. In vulnerable fast-flow aquifers, safe setback distances required are too large to practically achieve, thus high level of treatment, such as UV treatment, is required before land disposal of effluent. Together with considering site-specific conditions, the setback distances estimated in this study can be used to guide decision making in rural development and planning.

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