

# Microbial contamination in groundwater supply in a cold climate and coarse soil: case study of norovirus outbreak at Lake Mývatn, Iceland

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## ABSTRACT

This paper explores the fate and transport of microbial contamination in a cold climate and coarse aquifers. A confirmed norovirus outbreak in a small rural water supply in the late summer of 2004, which is estimated to have infected over 100 people, is used as a case study. A septic system, 80 m upstream of the water intake, is considered to have contaminated drinking water. Water samples tested were negative for coliform and strongly positive for norovirus. Modelling predicts that a  $4.8\text{-log}_{10}$  removal was possible in the 8 m thick vadose zone, while only a  $0.7\text{-log}_{10}$  and  $2.7\text{-log}_{10}$  removal in the aquifer for viruses and *Escherichia coli*, respectively. The model results support that the 80 m setback distance was inadequate and roughly 900 m aquifer transport distance was needed to achieve  $9\text{-log}_{10}$  viral removal. Sensitivity analysis showed that the most influential parameters on model transport removal rate are grain size diameter and groundwater velocity, temperature and acidity. The results demonstrate a need for systematic evaluation of septic systems in rural areas in lesser studied coarse strata at low temperatures, thereby strengthening data used for regulatory requirements for more confident determination on safe setback distances.

**Key words** | coarse soil, groundwater, microbial transport, norovirus outbreak, septic system

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## INTRODUCTION

Availability and access to safe drinking water are critical components of public health. Groundwater resources are generally considered to be the safest for drinking water supplies because of the protected layer of soil above the aquifer. The soil has a natural ability to filtrate out water pollution and therefore disinfection is generally not conducted in groundwater supplies serving rural communities. Yet, groundwater resources are vulnerable to sewage pollution, stemming from septic tanks, broken sewer lines and land application of sewage effluent (Woessner *et al.* 2001; Kvitsand & Fiksdal 2010). Sewage contains pathogens and increases nutrients as nitrate in groundwater which can take decades to dissipate (Pedley *et al.* 2006; Dunn *et al.* 2012).

Drinking water contamination, leading to waterborne diseases, is a recurrent event worldwide. A recent study

established that more than one out of every three waterborne outbreaks in affluent nations was caused by sewage contamination in groundwater (Hrudey & Hrudey 2004, 2007). Generally, multiple mechanisms were found to have contributed to the outbreaks and adverse conditions had often been in place for a long time (Hrudey & Hrudey 2007). In addition, evidence of sporadic incidence of waterborne diseases is also appearing (Payment *et al.* 1997; Payment & Hunter 2001; Calderon & Craun 2006; Colford *et al.* 2006; Craun *et al.* 2006).

Half of the world's population lives in rural areas and many rely on septic systems (WHO/UNICEF 2010). In the USA, over 20% of households are served by septic systems (Motz *et al.* 2012) and in Europe around 30% of the population lives in rural areas and many use septic systems for disposal of wastewater effluent (WHO/UNICEF 2010). At

the same time, 10% of Europeans rely on a small or very small water supply for drinking water (Hulsmann 2005). This widespread use of septic tanks can pose a significant threat to groundwater supplies. This risk is especially great in rural communities, which rely on untreated groundwater for their drinking water supply. Therefore, it is imperative to protect groundwater resources and provide easily adapted guidelines for local rural communities, such as safe setback distances. However, this approach is not without challenges, as the determination of safe setback distances requires a thorough knowledge of local strata and groundwater properties.

Many factors are known to influence the fate and transport of microorganisms in groundwater aquifers. A recent literature review suggests that pumice sand may be the most efficient soil type in removing microorganisms (Pang 2009). Specifically, the low pH often present in such soils and high surface area contribute to the sorption of microbes to the solids. Pang *et al.* (2003) concluded from laboratory experiments and groundwater modelling that a 48 m setback distance was enough to meet the Drinking Water Standards of New Zealand 2000 for enteric viruses in pumice sand aquifers (pH ~ 7) with groundwater speeds of <7 m/day. This distance was estimated to allow for 10-log<sub>10</sub> removal of viruses. However, this setback distance was established in uncontaminated aquifers. Wall *et al.* (2008) suggest that viral removal may be significantly lower in contaminated compared with uncontaminated pumice sand aquifers, leading to greater setback distances. Furthermore, viruses are known to be highly persistent and travel long distances in groundwater, and more so in cold water (Yates *et al.* 1985; Pedley *et al.* 2006; Borchardt *et al.* 2011; WHO 2011).

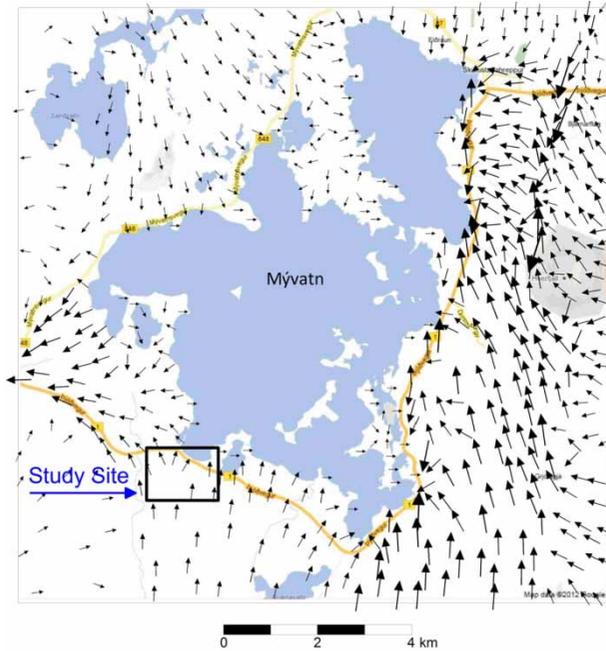
Until recently, limited research has been conducted on microbial transport in cold water in highly permeable coarse aquifers, although such conditions are common (DeBorde *et al.* 1999; Woessner *et al.* 2001; Kvitsand & Fiksdal 2010). Icelandic water supplies provide a good basis for such studies, both because they serve 95% of the population and many of them are located in the active volcanic zone with basaltic lava with high permeability (Sigurdsson & Sigbjarnarson 1985) and temperatures are usually between 3 and 6 °C (Sigurdsson & Einarsson 1988). Groundwater is not treated unless there is a danger of surface water intrusion. Ultraviolet (UV) irradiation treatment

together with filtration is practiced in Iceland, while residual disinfection is not (Gunnarsdottir *et al.* 2012b). Although Iceland is a sparsely populated country and the water supplies are generally considered safe, twelve confirmed waterborne disease outbreaks have occurred in the last three decades, all at small water utilities, of which six were due to *Campylobacter* and six to norovirus (Geirdottir 2011). The last confirmed outbreak was in 2004 and at least one contamination event has been confirmed since 2004, but was not associated with adverse health impacts (HAUST 2010). Some of the largest of these outbreaks were in groundwater supply systems where contamination originated from septic systems.

The goal of this research was to explore the fate and transport of microbial contamination in a cold climate and coarse aquifers. The 2004 norovirus outbreak in the rural Lake Mývatn area, which involved a large number of disease cases and the first time norovirus was detected in drinking water in Iceland (Atladdottir 2006), was used as a case study. A thorough literature search on the local conditions at Lake Mývatn, combined with groundwater model simulations, was used to explain why the outbreak occurred. Model results were compared to observed viral removal rates from a collection of aquifers with different site-specific properties. A sensitivity analysis on major model input parameters was performed to investigate what factors contributed to the occurrence (and timing) of the outbreak, and what factors would make water supplies especially vulnerable for viral outbreaks. Lastly, implications on regulatory environments are briefly discussed.

## LAKE MÝVATN SITE

Lake Mývatn (36.5 km<sup>2</sup>) is a protected nature reserve and one of Iceland's most popular tourist destinations. The lake is situated in the neovolcanic zone in Northern Iceland (65°35'), with geological formations from the last ice age (Pleistocene) and postglacial times. The area surrounding the lake includes groups of pseudocraters formed through steam explosions when lava plunged into the lake about 2,300 years ago (Thorarinsson 1979; Saemundsson 1991). The lake is predominantly groundwater fed (Figure 1) with moderately warm subsurface springs entering the lake on



**Figure 1** | Lake Mývatn area and the location of the study site. The arrows show the direction and magnitude of the groundwater flow according to the groundwater model by Vatnaskil (2007).

the eastern side and cold springs on the southern side (Olafsson 1979).

The study site, marked in Figure 1 and shown in Figure 2, is located in one of a group of pseudocraters on the south shore of the lake. The soils are heterogeneous permeable pumice. The mean particulate size diameter in four pseudocraters 4 km northeast of the study site was  $d_{50} = 8.3$  (4.7–13) mm and  $d_{10} = 1.05$  (0.8–1.4) mm with porosity of 42%. The soils are poorly to very poorly sorted, from medium gravel to sandy fine gravel (Dolvik & Höskuldson



**Figure 2** | Study site. Map of the study area (adapted from Jonsson (2006)). The water well and the septic tank are marked. The hotel and six dwelling houses are served by the well, four at Álftagerði and two close to the hotel.

unpublished data). The 5–10 m thick unconfined aquifer has a transmissivity of  $0.25 \text{ m}^2/\text{s}$  and 7 m/day seepage velocity, established from groundwater modelling (Vatnaskil 2007). The groundwater temperature is  $6^\circ\text{C}$  and basaltic, with a pH of 8.8, determined at a well 4.5 km east of the study site (Kristmannsdottir & Armannsson 2004).

A water well, 1.2 m deep, was installed in the 1960s, approximately 3–4 m from the lake shore (Figure 2), directly in the path of a large volume groundwater stream that flows into the lake from the south. A plastic barrier was installed between the lake and the well to prevent lake water from penetrating into the well. In addition, a concrete coating was constructed around and on the edges of the well (Geirsson 2007, 2010). The well supplies water to a seasonal summer hotel and six dwelling houses west of the hotel at Álftagerði, connected to the well with separate pipes (Figure 2). In 1996, a 20,000 L three chamber septic tank, with a 20 m drainage bed, was installed 80 m directly upstream of the water well (Sigmundsson 2008; Björnsson 2010). The septic tank was located in an area on a sill with limited vegetated cover. A sharp 8–9 m vertical drop in land elevation occurs between the sill and the lake, indicating a minimum 8 m vertical separation between the disposal depth and the groundwater table.

## THE 2004 WATERBORNE OUTBREAK AT LAKE MÝVATN

In the beginning of August 2004, an outbreak of gastrointestinal illness was reported by a group of tourists travelling in an organized bus tour around Iceland. The group had dined at a hotel on the south shore of Lake Mývatn in the evening of July 31 (Figure 2). The first case of illness was reported in the evening of August 1, when the group was in the nearby town of Akureyri in Northern Iceland. The group consisted of 26 individuals of whom 21 became ill. In the period of July 31 to August 3, individuals from three other tourist groups dining at the same hotel were reported ill (Atladdottir 2006). Simultaneously, residents in nearby summer houses were reported ill. An advisory to boil drinking water was issued on August 4 after which no case of illness was reported. It is estimated that at least 100 people became infected that summer from that same water supply.

A norovirus outbreak had occurred at the same hotel in late summer 2001 when at least 117 people became ill and food contamination was suspected as the culprit, but was later recognized to have had the same cause as the outbreak in 2004 (Briem 2005). Local residents also reported illness and that illness was a reoccurring event in late summer (Atladdottir 2006). However, when water analysis in 2004 showed that drinking water at the Lake Mývatn hotel was contaminated with the same genogroup (genogroup II) of norovirus as was found in patient stools, the outbreak was confirmed as waterborne. The owner of the hotel was requested to make the necessary improvement to the water supply. The following spring UV treatment was installed and drainage from the septic tank was moved away from the direction of the groundwater stream (Brynjolfsson 2008).

### Bacteriological testing results

Water samples were taken by the Local Competent Authority to identify the source of the 2004 outbreak. Three samples for bacteriological testing were taken on August 4, one from the hotel tap, one from a dwelling house and one from the lake near the well. They were analysed by the laboratory of the Environmental Agency of Iceland for HPC (Heterotrophic Plate Counts) at 37 and 22 °C, total coliform, faecal coliform (if coliform was found), *Salmonella* and *Campylobacter*.

The results of bacteriological testing are displayed in Table 1, along with analysis of routine samples taken in the spring of 2004 before the outbreak, and in summer

2005, nearly a year after the outbreak and when mitigation measures had been taken. For comparison, the values in the last line of the table represent the water quality limits set by the Icelandic Drinking Water Regulation (IDWR). All water samples from the water supply satisfied bacteriological requirements for IDWR except samples taken in the drinking water well before treatment, a year after the outbreak, where the HPC was just above the IDWR limits and turbidity was also higher than usual, although below the limit. This indicates that some organic contamination was present in the groundwater, but this was successfully UV treated before being supplied to users.

### Viral testing results

Five stool samples were taken from people reported sick and were tested for viral contamination at the University Hospital Laboratory, four of which were found to be positive for norovirus of genogroup II (Atladdottir 2006).

Seven water samples were taken and tested for norovirus at the Firrst Life Science Laboratory in Finland (Atladdottir 2006): First, two samples were taken on August 4; one from the tap at the hotel and one from a summerhouse in the neighbourhood where illness had been reported. Twelve days later, August 16, five samples were collected; one from each tap, the same as on August 4, and three from Lake Mývatn near the well.

The results of the viral testing, shown in Table 2, demonstrate that norovirus was present in the drinking water of the same genogroup as was found in stool from patients.

**Table 1** | Results from general water sample monitoring 2004–2005 at Lake Mývatn (The Environmental Agency of Iceland 2004, 2005)

Sampling date	Sample site	Turbidity NTU	Conductivity $\mu\text{S}/\text{cm}$	HPC at 37 °C in 1 mL	HPC at 22 °C in 1 mL	Coliforms in 100 mL	Faecal coliform in 100 mL	<i>Salmonella</i> in 400 mL	<i>Campylobacter</i> in 400 mL
25.5.2004	Hotel tap – routine inspection	0.1	190	N.D.	19	0	N.D.	N.D.	N.D.
4.8.2004	Hotel tap	N.D.	N.D.	1	25	0	N.D.	Neg.	Neg.
4.8.2004	Dwelling house	N.D.	N.D.	0	33	0	N.D.	Neg.	Neg.
4.8.2004	Lake near well	N.D.	N.D.	990	2,100	990	990	Neg.	Pos.
4.7.2005	Well (untreated)	0.27	190	N.D.	110	0	N.D.	N.D.	N.D.
4.7.2005	Hotel tap (treated)	<0.1	190	N.D.	9	0	N.D.	N.D.	N.D.
	IDWR	<1.0	<2,500	N.R.	<100/mL	0/100 mL	0/100 mL	0	0

N.D. = not done. Test for faecal coliform is not done if coliform is not detected.

N.R. = no requirements.

**Table 2** | Results from norovirus tests of water samples taken August 4 and 16, 2004 at Lake Mývatn (Firrst Life Science 2004)

Water sampling site	Water samples from August 4, 2004	Water samples from August 16, 2004
Network – hotel tap	Very strong positive – (genogroup II)	Positive – (genogroup II)
Network – private house tap	Very strong positive – (genogroup II)	Negative
Lake/A202	N.D.	Negative
Lake/A203	N.D.	Negative
Lake/A204	N.D.	Negative

N.D. = not done.

The samples registered as very strongly positive on August 4 and positive on August 16. Norovirus was found both at the hotel tap and at the dwelling house on August 4, but only at the hotel on August 16. The dwelling houses are connected to the well but with separate pipes. Samples from the lake tested negative for norovirus. This indicates that the water well was contaminated by the sewage from the upstream septic tank and not the lake, although virus could have been too diluted in the lake for detection.

## MICROBIAL TRANSPORT MODEL

### Simple transport model

Microbial removal rates in the unsaturated and saturated zones are often described by the means of simple transport models (Pang 2009). For a continuous release of sewage, microbial concentration will ultimately reach a steady state, which represents the highest possible microbial content downstream of the point of contamination release. Neglecting dispersion and dilution, the governing equation for microbial transport in groundwater with kinetic sorption is

$$u \frac{\partial C}{\partial x} = -\lambda C \quad (1)$$

The term on the left hand side represents transport via advection, where  $u$  is the groundwater seepage velocity. The term on the right hand side combines the removal associated with inactivation of free and sorbed microorganisms,  $\mu_1$

and  $\mu_s$  respectively, as well as the attachment  $k_{att}$  and detachment  $k_{det}$  of microorganisms on solid strata, i.e.

$$\lambda = \mu_1 + k_{att} \left( 1 - \frac{k_{det}}{k_{det} + \mu_s} \right) \quad (2)$$

Equation (2) suggests that total removal rates are bounded on one hand by the free microbes inactivation rate, i.e.  $\lambda = \mu_1$  if  $k_{det} \gg \mu_s$ . On the other hand, if detachment rates are slow ( $k_{det} < \mu_s$ ), as suggested by field and modelling studies in dune sand, (e.g. Schijven *et al.* 1999, 2006), Equation (2) reduces to

$$\lambda = \mu_1 + k_{att} \quad (3)$$

This scenario represents the maximum removal due to sorption. The solution of Equation (1) is an exponential decay of groundwater contamination with distance from source  $x$ , from which the  $\log_{10}$  removal rate is determined as

$$\log_{10} \left( \frac{C}{C_0} \right) = -\frac{\lambda}{u \cdot 2.3} x \quad (4)$$

The slope of the curve  $\lambda/(u \cdot 2.3)$  is referred to as the total  $\log_{10}$  removal rate and is expressed in the unit of  $\log_{10}/m$ . If this removal rate is a constant, independent of  $x$ , the removal is linear. Pang (2009) found that 70% of the 87 datasets investigated were better described by a linear law and 30% with a power law, implying reduced removal rate with distance. Equation (4) is generally used to describe microbial removal, both in the vadose zone as well as in aquifers.

Equation (4) implies that safe setback distance is linearly correlated with  $\log_{10}$  removal and inversely correlated with removal rate  $\lambda/(u \cdot 2.3)$ . As an example, for 9- $\log_{10}$  removal requirement, Equation (4) yields the safe setback distance

$$X_{9-\log} = \frac{9}{\left( \frac{\lambda}{u \cdot 2.3} \right)} \quad (5)$$

### Sorption-filtration within groundwater aquifers

Sorption is the process whereby chemicals or organisms become attached to and detached from rock material. If

detachment rates are small, the dominant process is that of irreversible sorption, also referred to as filtration. Filtration theory for colloids in packed beds suggests that the attachment rate constant  $k_{\text{att}}$  can be described based on soil and microorganism properties, according to Harvey & Garabedian (1991), as

$$k_{\text{att}} = 3 \frac{(1-n)}{2d_c} \alpha \eta u \quad (6)$$

where  $d_c$  represents the average soil diameter of the single collector,  $n$  the porosity,  $u$  the groundwater seepage velocity,  $\eta$  the single collector efficiency and  $\alpha$  collision efficiency. Pang et al. (2005) and Harvey & Garabedian (1991) suggest using  $d_{10}$  instead of  $d_{50}$  as effective particle size  $d_c$  when variation in grain size is large.

The single collector efficiency is found to depend on three different mechanisms: Brownian diffusion, interception and sedimentation. For viruses, Brownian diffusion is found to dominate (Penrod et al. 1996), simplifying the attachment rate equation to

$$k_{\text{att}} = 6 \frac{(1-n)}{d_c} \alpha A_s^{1/3} \left( \frac{D_{\text{BM}}}{d_c n u} \right)^{2/3} u \quad (7)$$

Here  $A_s = 2(1-\gamma^5)/(2-3\gamma+3\gamma^5-2\gamma^6)$  is Happel's porosity dependent parameter with  $\gamma = (1-n)^{1/3}$ . The molecular diffusion coefficient  $D_{\text{BM}} = K_B(T+273)/(3\pi d_p \mu)$  is based on water temperature  $T$ , Boltzmann constant  $K_B$ , the diameter of viruses  $d_p$ , and the dynamic viscosity of water  $\mu$ , as described by Schijven et al. (2006).

MS2 bacteriophages have a similar size ( $d_p = 26$  nm) as noroviruses and are commonly used to represent norovirus sorption (Penrod et al. 1996; Schijven et al. 2006). Studies suggest that the collision efficiency of MS2 is affected by the pH of the groundwater. When water pH is below the isoelectric point of the virus and porous medium, the electrostatic attraction between the virus and opposite charge porous media promotes adsorption (Guan et al. 2003). Within the pH range of 3.5–7, Schijven & Hassanizadeh (2002) found that the following empirical relationship applies:

$$\alpha = \alpha_0 0.9 \left( \frac{\text{pH} - \text{pH}_0}{0.1} \right) \quad (8)$$

The collision efficiency  $\alpha$  is generally back calculated from tracer experiments in the field. In the absence of tracer experiments, the reference values for the Lake Mývatn study area were chosen from previously published studies with similar groundwater and strata properties. In particular, basaltic aquifers ( $\text{pH} > 7$ ) were chosen in order to eliminate the influence of pH, given that Equation (8) is only valid for acidic environments. In addition, the selection criteria included sufficient groundwater speeds ( $u > 1$  m/day) and lateral distances ( $x > 30$  m). For norovirus modelling,  $\alpha = 2.7 \times 10^{-4}$  was chosen based on Schijven et al.'s (1999) MS2 tracer experiments in a contaminated sand dune ( $d_c = 0.2$  mm,  $u = 1.2$  m/day) aquifer with similar pH (i.e.  $\text{pH} = 7.8$ , range 7.3–8.3) and distance ( $x = 30$  m). This corresponds to a conservative value for collision efficiencies in coarse alluvial gravel aquifers (Pang et al. 2005) and accounts for contamination build-up in the aquifer which may undermine sorption, according to Wall et al. (2008). For *Escherichia coli* modelling,  $\alpha_0 = 4.5 \times 10^{-3}$  based on field experiments by Mutsvangwa et al. (2006) with the same bacteria group in a sand aquifer ( $d_c = 0.7$  mm,  $u = 1.3$  m/day,  $\text{pH} = 8.5$ ,  $x = 500$  m).

Accounting for the grain diameter ( $d_{10} = 1.05$  mm) in the poorly sorted strata and groundwater seepage velocity ( $u = 7$  m/day) at Lake Mývatn, Equation (7) yields  $k_{\text{att}} = 0.06$  day<sup>-1</sup> for MS2 transport. For *E. coli*, the single collector efficiency was found to be dominated by Brownian diffusion based on the corrected Rajagopalan & Tien (1976) version as presented by Mutsvangwa et al. (2006) and Logan et al. (1995). The attachment rate for *E. coli* ( $d_p = 0.5$  μm) was estimated as  $k_{\text{att}} = 0.14$  day<sup>-1</sup> based on Equation (7).

## Inactivation

The inactivation rate,  $\mu_1$ , of free pathogens is related to many physical and chemical factors, temperature of the water being one of the most important. The inactivation rate for viruses can be one order of magnitude higher at 25 °C than at 5 °C (Pedley et al. 2006). The temperature of the spring water south of Lake Mývatn is about 6 °C and is independent of season (Olafsson 1979; Kristmannsdottir & Armannsson 2004).

MS2 bacteriophages have been found to be good surrogates for norovirus inactivation (Collins et al. 2006; Bae & Schwab 2008). Yates et al. (1985) measured the free

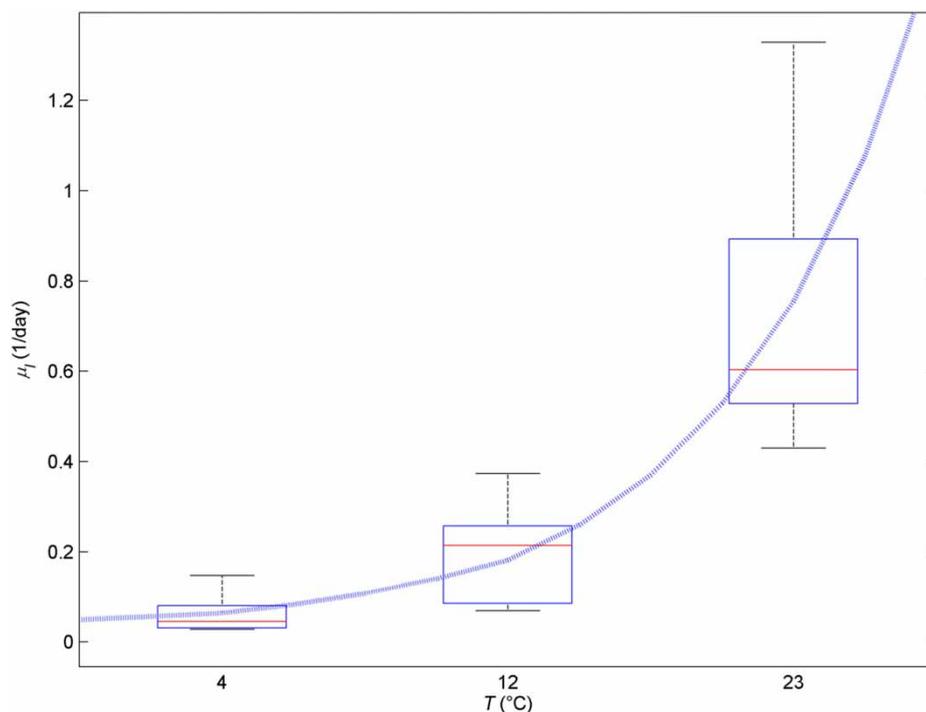
inactivation rate of MS2 in five different groundwater aquifers at three different temperatures, 4, 12 and 23 °C. These data, plotted in Figure 3, show a clear dependency on temperature, but also a significant spread between different groundwater aquifers, implying that site specific conditions may play an important role. The mean die-off values at each temperature were fitted with a log relationship, in order to account for the levelling of die-off rates at low temperatures. The fit yields a free inactivation rate of 0.08 day<sup>-1</sup> for the 6 °C cold groundwater at the Lake Mývatn site, in line with, for example, the 0.083 day<sup>-1</sup> at 5 °C used by Schijven *et al.* (2002).

Inactivation rates for the bacteria *E. coli* are estimated to be 0.4 day<sup>-1</sup>, using the mean rates from Pedley *et al.* (2006), as the limited data for *E. coli* do not show dependency on temperature.

### Log removal rates in the vadose zone

Few studies have been undertaken to assess the microbial removal occurring within the unsaturated vadose zone as opposed to the saturated groundwater. Pang (2009)

summarizes and compares removal rates of MS2 bacteriophages and faecal coliforms in the vadose zone from various studies. She argues that microbial removal rates for viruses (and virus indicators/phages) appear to be of the same order as for bacteria in the same soil media. In addition, microbial removal rates appear to increase with infiltration rates. The sewage effluent released to the hotel septic tank at the Lake Mývatn site is estimated as 35 m<sup>3</sup>/day based on standard usage guidelines in Iceland and the number of residents (Environmental Agency of Iceland 2004), which corresponds to a hydraulic loading rate of approximately 1 m/day. For a similar hydraulic loading rate, Gerba *et al.* (1991) found a MS2 removal rate of 0.53-log<sub>10</sub>/m in a vadose zone composed of sandy gravel and coarse sand. The same rate was found for faecal coliforms, representing bacterial removal in a 3 m thick sand vadose zone with varying hydraulic loading. Sinton *et al.* (2000) studied septic tank effluent in coarse gravels and found that faecal coliform removal rates ranged from 0.27 to 0.5 log<sub>10</sub>/m, with a mean of 0.44-log<sub>10</sub>/m. Hence, a representative removal rate for both viruses and bacteria within the vadose zones of coarse gravel and sand aquifers is



**Figure 3** | Free inactivation rate of MS2 as a function of groundwater temperature based on experiments from Yates *et al.* (1985). The central mark is the median, the edges of the box are the 25th and 75th percentiles, and the whiskers extend to the most extreme data points not considered outliers. The dotted line represents the best log fit through the data,  $\mu_i = 0.0384e^{0.1295T}$ .

within  $0.44\text{--}0.53\text{-log}_{10}/\text{m}$ , which was used as a base for the Lake Mývatn study site.

### Microbial removal requirements for safe drinking water

Since the IDWR does not specify any requirement for viruses, the drinking water requirements in other countries were consulted. The Drinking-Water Standards for New Zealand 2000 (DWSNZ) require less than 1 per 100 L for enteric virus, corresponding to a  $10\text{-log}_{10}$  removal (Pang *et al.* 2003). Alternatively, the requirements used in a recent Dutch study are  $9\text{-log}_{10}$  removal (Schijven *et al.* 2006). For the present study, a  $9\text{-log}_{10}$  removal was used as a minimum requirement for enteric viruses.

The IDWR for *E. coli* is zero in 100 mL (Table 1). Medema *et al.* (2003) reported a typical *E. coli* concentration on the order of  $10^5\text{--}10^7$  n/100 mL;  $10^7$  was used in this study. This means that, in order to satisfy the IDWR, a minimum  $7\text{-log}_{10}$  removal is required for *E. coli*.

## RESULTS AND DISCUSSION

### Removal at the Lake Mývatn groundwater well

The groundwater transport model with the site-specific conditions, discussed above under Microbial transport model, suggests that a  $3.5\text{-log}_{10}$  to  $4.8\text{-log}_{10}$  viral and bacterial removal may be possible within the 8 m thick vadose zone at Lake Mývatn, corresponding to observed log removal rates of  $0.44\text{--}0.53\text{-log}_{10}/\text{m}$  for coarse sand and gravel media (Pang 2009). Within the 80 m lateral distance travelled in the saturated zone, between the sewage discharge point and the drinking water well, however, the model estimates a  $0.7\text{-log}_{10}$  removal for MS2, representing norovirus, and a  $2.7\text{-log}_{10}$  removal for *E. coli*. The modelled removal within the groundwater aquifer accounts only for a small portion of that achieved in the vadose zone.

Combined, the removal of viruses after infiltrating the vadose zone and travelling within the groundwater to the well is estimated at best as  $5.5\text{-log}_{10}$ , which does not satisfy the minimum  $9\text{-log}_{10}$  removal for safe drinking water. However, the combined removal of *E. coli* from the drinking water is estimated as  $7.5\text{-log}_{10}$ , which conforms to the

minimum  $7\text{-log}_{10}$  bacteria removal discussed above under Microbial removal requirements for safe drinking water. Therefore, the simple groundwater model adapted to the Lake Mývatn site supports the observation during the outbreak of drinking water strongly positive for norovirus (Table 2), though bacteria free (Table 1), indicating that the 80 m setback distance was insufficient.

### Comparison to observed viral removal rates

Table 3 compares the simulated MS2 removal rates at Lake Mývatn with observed removal rates at different sites with various groundwater strata and water properties, as summarized in Pang (2009) and references used in that paper. The safe setback distances, derived from Equation (5), represent solely the viral removal within the different aquifers and neglect removal in the vadose zone. The field observations show a clear dependency of groundwater log removal, and hence safe setback distances, on soil type: the safe setback distance for  $9\text{-log}_{10}$  removal in sand aquifers, with  $d_{50}$  smaller than 0.4 mm, is less than 50 m. This same distance is, however, on the order of several hundred meters in more coarse strata (sandy gravel, sand and gravel), with  $d_{50}$  exceeding 5 mm. The model prediction for coarse gravel pumice at Lake Mývatn, top row in Table 3, conforms to these field studies in that it predicts low viral removal. The modelled removal rate of  $0.009\text{-log}_{10}/\text{m}$  is, however, on the order of two to three times lower than that observed in the coarse gravel aquifer studied by Sinton *et al.* (2000). The derived safe setback distance is slightly less than 1 km as opposed to several hundreds of meters. This difference cannot be entirely explained by the different groundwater temperatures and pH because these effects are counterbalanced by the different groundwater seepage velocities. Hence, this may be an indication that the model predicts conservative removal rates, which can be explained in several of the underlying model simplifications. The sorption module, for example, does not account for specific features of pumice strata, such as their high surface areas which promote removal (Pang 2009). The model excludes the dispersion of pollutants and dilution of fresh water in the well. Lastly, the uncertainty in model input data may play a role as well, which will be explored further in the following section.

**Table 3** | Modelled MS2 removal rates at Lake Mývatn and previous field observations in groundwater aquifers. Adapted and expanded from Pang (2009)

Location	Aquifer	d (mm)	n	u (m/day)	T (°C)	pH	x (m)	Removal rate (log <sub>10</sub> /m)	x <sub>9-log</sub> (m)	Reference
Mývatn, Iceland	Gravel pumice	8.26 (d <sub>50</sub> ) 1.05 (d <sub>10</sub> )	0.42	7	6	8.8	80	0.009	960	
Castricum, Netherlands	Dune sand	0.2–0.24 (d <sub>gm</sub> )	0.35	1.2–1.7	2–5	7.3–8.3	30	0.187	48	Schijven et al. (1999)
Netherlands	Coarse sand	0.4	0.32	0.33–0.56	NA	7.5	37.7	0.188	48	van der Wielen et al. (2006)
Rotorua, New Zealand	Pumice sand	0.15 (d <sub>50</sub> )	0.2	0.9–1	13	6.2	2	1.85	5	Wall et al. (2008)
Montana, USA Frenshottown High School	Sand and gravel	2.4 (d <sub>mean</sub> )	0.2	1–2.9	9–12	6–6.4	6.6–17.4	0.392	23	Deborde et al. (1998)
Montana, USA Erskine Fish Access	Sandy gravel	1.25 & 12 (d <sub>mean</sub> )	0.15	22–30	10.3	7.2	7.5–40.5	0.0994	91	Deborde et al. (1999)
Montana, USA Erskine Fish Access	Sand and gravel	1.25 & 12 (d <sub>mean</sub> )	0.15	129	10.3	7.2	21.5	0.0358	250	Woessner et al. (2001)
Burnham, New Zealand	Coarse gravel	18 (d <sub>50</sub> ) 0.9 (d <sub>10</sub> )	0.2	88–112	12	6.92	87	0.025	360	Sinton et al. (2000)

NA, not available.

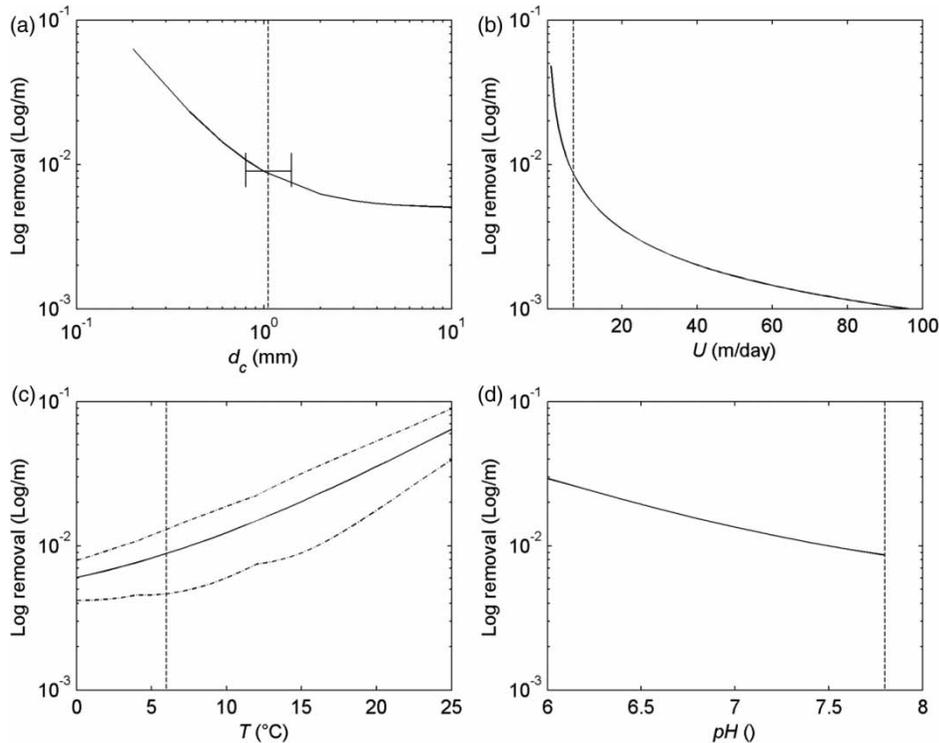
### Groundwater model sensitivity

The groundwater transport model is dependent on selected site specific model input parameters, including grain size, groundwater seepage velocity, temperature, pH and collision efficiency. Figure 4 presents the log removal rate,  $\lambda/(2.3 u)$  (Equation (4)), for a range of input variables, which was based on observed values in sand, sand and gravel, and coarse gravel aquifers (Table 3). The vertical lines represent the result of the Lake Mývatn study, with modelled MS2 removal rate of 0.009-log<sub>10</sub>/m.

First consider the model sensitivity on grain size diameter. Equation (7) shows that the attachment rate,  $k_{att}$ , scales as  $d_c^{-5/3}$ . This means that strata with a 10 times greater diameter may have approximately 50 times lower attachment rate and removal rate if attachment dominates inactivation,  $k_{att} \gg \mu_1$ . In most aquifers, die-off contributes substantially to viral removal, which in turn would moderate the impact of grain size. At Lake Mývatn, where  $k_{att} \approx \mu_1$ , the characteristic grain size  $d_{10}$  ranged from 0.8 to 1.4 mm in four different samples taken at the site (see horizontal line, Figure 4(a)). The model suggests that the removal rate may vary 20% from the mean, corresponding to 0.011 and 0.007-log<sub>10</sub>/m, respectively. While this is a significant range, it does not alter the previous result that 80 m travel distance was not sufficient to achieve a 9-log<sub>10</sub> removal at the Lake Mývatn drinking water well.

Next, consider the model dependency on groundwater seepage velocity. According to Equation (7),  $k_{att} \sim u^{1/3}$ , so the removal rate,  $\lambda/(u \cdot 2.3)$ , scales between  $u^{-2/3}$  if  $k_{att} \gg \mu_1$  and  $u^{-1}$  if  $k_{att} < \mu_1$ . This suggests, for example, that a 10-fold groundwater velocity may reduce the removal rate anywhere from five to 15 times, all other parameters being equal. Since coarse strata are typically characterized by large grain size and seepage velocity (Table 3), the combined effect would generally be additive. The model thus conforms with the field observations in Table 3, that removal rate is on the order of 10 times lower for coarse (gravel) than fine (sand) aquifers. At the Lake Mývatn site, Figure 4(b) shows that the log removal may vary 20–30% from the mean if the uncertainty in the seepage velocities were  $\pm 2$  m/day.

The model sensitivity to groundwater temperature is predominantly associated with the exponential dependency of



**Figure 4** | Model sensitivity for four parameters. Modelled log removal rate as a function of (a) grain size, (b) groundwater seepage velocities, (c) water temperature, and (d) pH. The vertical broken lines represent the base simulation for Lake Mývatn, and the dashed dotted lines in (c) ranges.

inactivation rate,  $\mu_1$ , on water temperature. Figure 3 shows, for example, that when groundwater temperature drops from 15 to 10 °C, the inactivation decreases by a factor of two. An additional drop to 5 °C decreases the die-off rate by a factor of 4. If viral inactivation dominates grain attachment, i.e.  $\mu_1 \gg k_{att}$ , the total log removal would be linearly correlated with inactivation. In such cases, groundwater temperature could greatly influence the log removal rate and consequently the safe setback distance. This strong influence of temperature has received limited attention in contamination studies, nor have many studies focused on a low temperature environment (John & Rose 2005).

Figure 4(c) portrays the potential influence of water temperature for strata at the Lake Mývatn site. The solid line represents the mean relationship, and the dot-dashed lines the range, derived from Yates *et al.*'s (1985) experiments on soils from five different aquifers, as portrayed in Figure 3. Figure 4(c) suggests that the removal rate in a cold climate like Iceland's, where groundwater temperature originating from melting glaciers can be as low as 2 °C (Adalsteinsson *et al.* 1992), may be 10 times lower than for similar strata

in Mediterranean climates, where temperatures may exceed 20 °C. Figure 4(c) also indicates that the log removal rate may vary from 0.005 to 0.013  $\log_{10}/m$  depending upon whether the upper or lower limits of Yates *et al.*'s (1985) data are used. This high uncertainty associated with site dependent characteristics demonstrates the need for conducting more microbial inactivation studies to better understand the role of local strata (other than temperature) on inactivation rate.

Lastly, the collision efficiency,  $\alpha$ , is generally derived from tracer experiments in the field and is affected by groundwater acidity. The reference for Lake Mývatn was taken from Schijven *et al.*'s (1999) study of basaltic contaminated dune sand aquifer. Figure 4(d) portrays the possible impact of groundwater acidity on a removal rate based on Equation (8). The figure suggests that neutral groundwater (pH = 7) might have a 60% higher log removal rate or 0.014- $\log_{10}/m$ . Another point of consideration is that increasing coarseness and water cleanliness may improve the collision efficiency, which in turn increases the viral removal rate. For example, Pang *et al.* (2005) derived

$\alpha_{\text{mean}} = 1.9 \times 10^{-5}$  based on six well tests performed by Sinton *et al.* (2000) in a neutral aquifer and clean, coarser gravel (Table 3) than at Lake Mývatn. This higher  $\alpha$  serves to counterbalance the impact of high seepage velocities ( $u > 80$  m/day, Figure 4(b)).

To conclude, the high sensitivity to four of the model input parameters (Figure 4) highlights the need for conducting field experiments to reduce the uncertainty of results and calibrate the groundwater model. It also indicates that coarse, permeable and cold climate groundwater environments may be especially susceptible to microbial contamination.

### Groundwater viral removal potential of gravel pumice and regulation implications for Iceland

Pang (2009) and Pang *et al.* (2003) argue that pumice sand may be the most efficient soil type in removing microorganisms. High surface areas and low pH contribute to the sorption of microbes to the solids. A safe setback distance of 48 m was established for enteric viruses in such pumice sand. A similar distance can be derived based on sand aquifer studies in the Netherlands (see Table 3).

The norovirus outbreak in Lake Mývatn, where a septic tank is located 80 m upstream of a drinking water well, is evidence that a 48 m setback distance is not sufficient for gravel pumice in a cold climate. A groundwater model incorporating general filtration theory and studies of inactivation rates of viruses suggests that a larger grain size, higher groundwater seepage velocity, cold, and basaltic groundwater may all contribute to undermine the removal rate in coarse gravel pumice. This may have significant implications for groundwater supplies in coarse strata and a cold climate. The greatest risk is likely to occur in small, and less regulated, rural water supplies, supported by the number of waterborne outbreaks reported in supplies in Iceland during recent decades. Many rural water systems serve a large number of tourists during the summer months, as well as farms producing agricultural products, yet few studies on the transport of microbes in cold coarse strata have been carried out. This highlights a need for research on hydraulic parameters and the travel of pathogens in coarse strata, both with respect to geological conditions and

temperatures, to underpin regulations governing determination of water protection zones for rural groundwater wells. Our initial effort suggests that the safe setback distances for achieving a 9- $\log_{10}$  viral removal might be up to 1 km for site specific conditions at Lake Mývatn (Table 3), neglecting initial removal in the vadose zone. This is more in line with 4 km safe setback distances for 7- $\log_{10}$  viral removal reported in alluvial gravel aquifers (Pang *et al.* 2005). Yet, with the data available today, it is impossible to assess whether the safe setback distance is indeed several hundreds of meters or up to or more than a kilometre.

Another question worth considering is what type of measure would be most appropriate for determining water protection zones. Table 3 lists safe setback distances, an approach taken in many countries. From Equations (3), (5) and (7), however, it can be seen that safe setback distances scale linearly on  $u$  if  $\mu_1 \gg k_{\text{att}}$  and  $u^{2/3}$  if  $\mu_1 \ll k_{\text{att}}$ . This undermines the use of setback distances for defining protection zones for different groundwater supplies. This dependency may be reduced by using travel times,  $X_{\log}/u$ , as a measure for the protection zone. The travel time between the septic tank and well at Lake Mývatn is estimated as 11 days, which is shorter than the 50 day travel zone used in some regulations, indicating that the setback distance is significantly too short.

Lastly, the severity of the Lake Mývatn outbreak discussed in this article and the inadequate setup of the septic system demonstrate a need for systematic review of existing septic systems in Iceland and comprehensive regulatory guidelines for installation of such systems. This could be included in a systematic preventive management system, such as a water safety plan, that has been or is currently being implemented by many utilities (Gunnarsdottir & Gissurarson 2008; Gunnarsdottir *et al.* 2012a). A possible outcome of a review would be installation of UV treatment where needed or even a reconfiguration of the septic system if the risk is deemed unacceptable.

### Factors contributing to the timing and occurrence of outbreaks

The contamination at the Lake Mývatn study site originated from a septic tank serving predominantly summer dwellings

and a hotel. The tourist season starts in late May or the beginning of June. Norovirus outbreaks in 2001 and 2004 were reported in late July and the beginning of August. In an interview, a summerhouse dweller claimed that illness was a recurrent event in late summer.

The late season timing of outbreaks may be explained by the experimental findings of Wall *et al.* (2008). The addition of dissolved organic carbon were found to progressively reduce removal and retardation of phages in saturated pumice sand aquifers, suggesting that less removal may be achieved in contaminated as opposed to uncontaminated aquifers. At Lake Mývatn, sewage contamination starts building up in the aquifer at the beginning of the tourist season. The outbreak timing, at the end of July, may indicate that a critical build-up of contamination is reached after roughly two months of operation.

Another factor known to contribute to increased microbial contamination is precipitation, which increases the soil saturation and enhances infiltration in the groundwater table. The removal capacity of the vadose zone is found to be inversely correlated with infiltration rates (Pang 2009). Waterborne outbreaks have been associated with extreme precipitation (Curriero *et al.* 2001; Taylor *et al.* 2004). The septic tank was present in an area with limited vegetative cover and pumice soils. Hence most of the rain infiltrates the ground and reduces the travel time in the vadose zone. However, the rain pattern in Iceland is generally characterized by events of low intensity and long duration. The rain record at the local meteorological station at Lake Mývatn indicates that the summer of 2004 was relatively dry (Gísladóttir 2007). A prolonged 3 day rain event with a maximum of 6 mm/day occurred 10 days prior to the reported cases of illnesses, which matches closely the travel time of 11 days. While it is possible that the rain may have accelerated the groundwater recharge, its intensity was much lower than the estimated sewage infiltration rate of 1 m/day. Rain may therefore have played a minor role in the occurrence of the outbreak. Peak occupancy at the hotel, and the fact that the septic tank at Lake Mývatn was inadequately sized according to design criteria given in the 2004 guidelines of the Environmental Agency of Iceland, probably played a larger role than the rain.

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## CONCLUSION

This study takes a first step in reviewing the potential of microbial contamination in groundwater supply in a cold climate and coarse soil. The sensitivity of microbial groundwater transport, explored by a model and tabulation of results from various studies, shows that microbial transport is particularly sensitive to temperature and grain size, and thus directly determines safe setback distances and the regulatory environment. These results were further corroborated by the case study of a documented waterborne norovirus outbreak at Lake Mývatn in Iceland. The model was applied to the site and results confirm field observations that an 80 m setback distance (11 day groundwater travel time) between a septic tank and a drinking water well was inadequate for achieving a 9-log<sub>10</sub> viral removal, but sufficient for a 7-log<sub>10</sub> bacterial removal. The model highlights the finding that aquifers with large grain size, high seepage velocity, cold temperatures and high pH contribute to adverse conditions for microbial removal. In addition, contamination build-up associated with seasonal septic tank discharge may play an important role in reducing the filtration capacity of the volcanic strata. The vadose zone contributes significantly to the initial removal of microbial contamination and needs to be considered in the transport model. These results highlight the need for further studies on microbial removal rates in saturated and unsaturated volcanic strata in a cold climate. Results from such studies should then be used to determine and reinforce regulations regarding safe setback distances for septic tanks in rural areas that take into account local hydrogeologic settings.

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