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ORIGINAL ARTICLE

An assessment of hydrologic factors that influence contaminant concentrations determined from domestic well-water samples

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Abstract

Open borehole wells in fractured crystalline rock are commonly used for domestic water supply. Concentrations of contaminants present in the well water, typically collected from the tap after some degree of purging, are weighted average concentrations. In this study, factors which influence sample concentrations collected from domestic wells are evaluated through the use of conceptual models and confirmatory field tests. Focus is on the extent of drawdown during sampling, distance between the pump inlet and contaminated fractures, relative fracture transmissivity, fracture discharge rate into the borehole, and the hydraulic head distribution of the fractures. Simplified spreadsheet models were developed to assess the relation between these factors and sample concentrations. The models show that in addition to fracture transmissivities, drawdown in relation to fracture head plays a significant role in how much water recharges a well from each fracture, yet it is rarely measured when sampling domestic wells. To evaluate the extent of drawdown that may occur under typical household usage, water levels were monitored in three domestic wells. Pumping tests were conducted in three wells to measure changes in concentration as a function of pumping duration. Downhole profiling was conducted to examine how pumping induces changes to water quality within the wellbore. The results indicate that these factors can have a strong influence on contaminant concentrations in samples collected from these wells. As such, the weighted averaging factors should be considered in sample collection, in data interpretation, in assessing exposure risk and in mapping contaminant distributions.

Keywords Fractured bedrock wells · Groundwater quality · Health risk assessment · Concentration averaging · Sampling methods · Domestic wells

Introduction

Open borehole wells in fractured crystalline bedrock are commonly used for domestic water supply. Contaminant concentration data derived from these wells are used for assessing potability, need for filtration, risk of exposure, and need for remedial measures. Past studies have shown that the water quality of a sample collected from a domestic well is influenced by factors that are not typically considered when interpreting concentration data. These factors include pumping rate, the amount of drawdown achieved in the well when sampling, pre-pumping concentration in the well between active fractures, duration of pumping, depth of pump inlet, and the characteristics of the fractures that provide water to the well such as contaminant concentration, depth, head, discharge and transmissivity (Brainerd and Robbins 2004; Elci et al. 2001; Libby and Robbins 2013; McMillan et. al. 2014; Sokol 1963; Tsang and Hale 1990). Studies have shown that flow-weighted averaging associated with these factors when

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collecting a sample can greatly distort actual concentration distributions and lead to highly misleading contaminant configurations (Martin-Hayden and Robbins 1997; Metcalf and Robbins 2007; Harte 2017). For example, a study by Molofsky et al. (2018) found variability in dissolved methane concentrations of up to 50% when continuously purging domestic wells. Shapiro (2002) demonstrates that when sampling multi-fractured open borehole wells, varying transmissivity and head distributions between fractures will lead to averaged concentrations that are not representative of the true groundwater chemistry. The recommended solution of that study was to sample discrete intervals with straddle packers; however, this solution cannot be used in domestic water supply wells with downhole pump systems in-place. In addition, such a solution can be very costly and does not provide a representative sample of what homeowners may be consuming.

Health-related water quality sampling typically entails the collection of a single sample after a short duration of purging (CT DPH 2009; RI DEM 2010; US EPA Region 1 2015; US EPA Region 8 2016; WHO 1997). Due to the influences of flow-weighted averaging, the water quality of such a sample may not represent the water quality throughout the range of household water usage. Though past studies have discussed the factors that influence flow-weighted averaging and how they may affect concentrations in open borehole wells, no practical solutions have been suggested to improve the collection of domestic well samples to better assess contaminant risk to water supply.

In this paper, conceptual models are used to explore how the above factors may influence concentrations determined in a sample collected from a domestic well in fractured crystalline bedrock. Field tests were used to confirm insights gained from the conceptual models. These included the following: the continuous monitoring of water levels in several domestic wells to evaluate ranges in drawdown obtained during domestic usage; conducting pumping tests in three wells with a history of elevated chloride levels to examine the variation in concentration as a function of purging time; and performing downhole water quality profiles before and after a pumping test to examine the variation of water quality with depth induced by pumping. The results of the models and field tests in this study are used to recommend improved sampling procedures for domestic wells.

Methods

Conceptual model of factors that influence concentration in the wellbore

The ambient hydraulic head within an open borehole well in fractured bedrock is a weighted average. This ambient hydraulic he is a function of the transmissivity and head of each individual water-bearing fracture which is defined in Eq. (1) (Sokol 1963):

$$h_w = \frac{h_1T_1 + h_2T_2 + \cdots h_iT_i}{T_1 + T_2 + \cdots T_i} \tag{1}$$

where h_w is the static hydraulic head in the wellbore, h_i is the head of fracture i, and T_i is the transmissivity of fracture i.

Similar to the hydraulic head in the well, the water quality within the wellbore at any time is a weighted average that depends on the water quality in each fracture and the discharge into the well from each fracture. The concentration in the well is defined by Eq. (2):

$$C_w = \frac{C_1Q_1 + C_2Q_2 + \cdots C_iQ_i}{Q_1 + Q_2 + \cdots Q_i} \tag{2}$$

where C_w is the weighted average concentration in the wellbore, C_i is the concentration in the water of fracture i, and Q_i is the discharge of fracture i into the wellbore. Figures 1

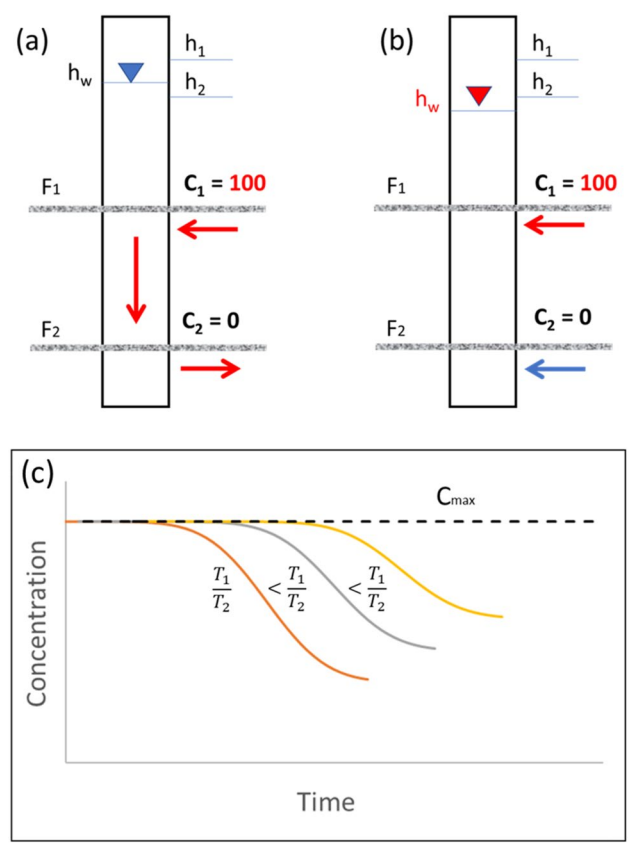

Fig. 1 Conceptual model illustrating wellbore concentration averaging effects during purging where the higher head fracture is contaminated. Parameters are shown for ambient conditions (**a**) and pumping conditions (**b**). As drawdown is achieved, average wellbore concentration changes with purging duration and is weighted towards the more transmissive fracture (**c**)

and 2 are simplified conceptual models which show a well with two fractures and illustrate how the drawdown of h_w can affect which fractures contribute water to the wellbore, the extent to which they contribute water, and the resulting weighted average concentration in the wellbore. Under ambient conditions in an open bedrock well with two transmissive fractures that have different hydraulic heads, water will flow into the wellbore from the fracture with higher head and exit the wellbore via the fracture with a lower head. During pumping conditions, the water column will undergo drawdown. If the drawdown is great enough to lower h_w to an elevation beneath the head in all fractures, water will flow into the well from all fractures.

Figure 1 illustrates a scenario where the elevation of fracture 1 (F_1) is higher than the elevation of fracture 2 (F_2), the hydraulic head in F_1 (h_1) is higher than the hydraulic head in F_2 (h_2), the contaminant concentration of the water in F_1 (C_1) is 100 (C_{max}) and the contaminant concentration of the water in F_2 (C_2) is 0 (presumes the contaminant will be diluted upon exiting the borehole). Under ambient conditions, water will always flow into the wellbore from F_1, and exit the wellbore through F_2 (Fig. 1a). When the well is fully flushed with water from fracture 1, C_w will equal the contaminant concentration level found in F_1.

Under pumping conditions in which drawdown occurs and h_w is below h_1 but above h_2, only F_1 contributes to the well and the C_w will equal the contaminant concentration level found in F_1. When the drawdown of h_w is below h_2, water will flow into the well from both F_1 and F_2 (Fig. 1b). In this case, C_w would be a flow-weighted average between the water from F_1 and F_2. When h_w is drawn down below h_2, C_w will enter a transient state as illustrated in Fig. 1c. As drawdown increases, the amount of water flowing into the well from F_2 increases relative to F_1, allowing water with $C=0$ to mix with the water in the wellbore causing C_w to decline. When a constant drawdown in the well occurs, C_w stabilizes. The magnitude and rate that C_w will decrease is influenced by the ratio of the transmissivities between F_1 and F_2 as well as the drawdown achieved relative to the fracture heads. In Fig. 1c, three curves are shown to illustrate how the transmissivity ratio comes into effect where from left to right: $\frac{T_1}{T_2} < \frac{T_1}{T_2} < \frac{T_1}{T_2}$. The higher T_2 is relative to T_1, the faster and greater the change in C_w.

Figure 2 illustrates an alternate case to Fig. 1 where the concentrations of the fractures are reversed with $C_1=0$, and $C_2=100$ (C_{max}). In this case, under ambient conditions when the well is fully flushed with water from F_1, $Cw=0$ given that $C_1=0$, and $h_1>h_2$. As h_w is drawn down and F_2 begins contributing water to the wellbore, the C_w begins to rise. As in Fig. 1, the magnitude and rate that C_w will increase is influenced by the ratio of the transmissivities between F_1 and F_2 as well as the drawdown. In this case the effect on C_w is the opposite of that shown in Fig. 1.

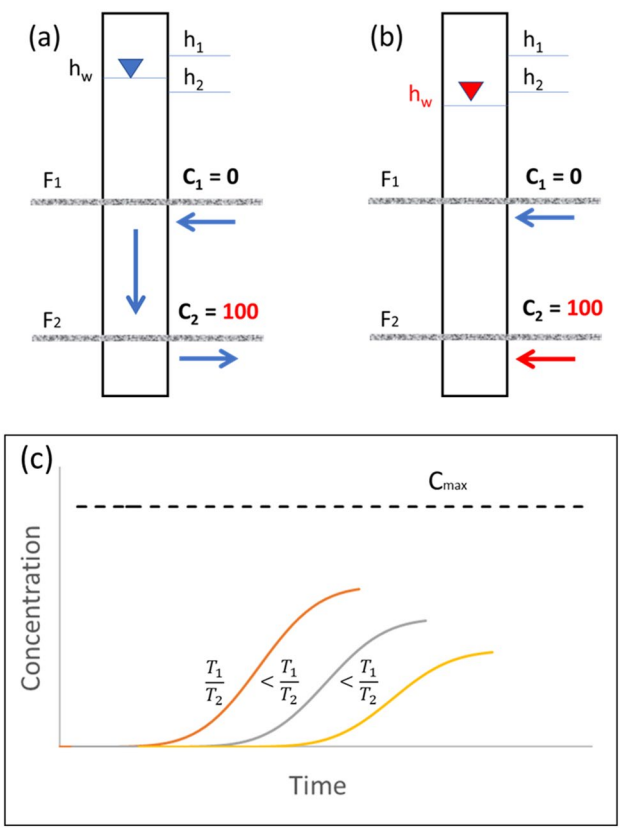

Fig. 2 Conceptual model illustrating wellbore concentration averaging effects during purging where the lower head fracture is contaminated. Parameters are shown for ambient conditions (**a**) and pumping conditions (**b**). As drawdown is achieved, average wellbore concentration changes with purging duration and is weighted towards the more transmissive fracture (**c**)

In-well concentration changes as a function of drawdown

To further examine how the water quality in the wellbore may change as drawdown is changing in Figs. 1 and 2, a simplified drawdown model was created in Microsoft Excel. The model uses the Theis equation to calculate the relationship between drawdown and discharge into a wellbore from two separate fractures with different hydraulic heads, transmissivities, and concentration levels. The model assumes that one fracture contains a contaminant and the second fracture does not. The model also assumes that steady state has been reached for each unit drawdown and that the water column is evenly mixed. Drawdown (s) intervals were predefined throughout a range of 0–10 m so that the Q of each fracture could be solved for throughout the range of drawdown values. The drawdown of each fracture was calculated by taking the difference between the head in the fracture and the head in the well at a predefined

drawdown in the wellbore. Q for each fracture is determined by rearranging the Theis equation as:

$$Q = \frac{4s\pi T}{W(u)} \tag{3}$$

The Q and C for each fracture are then used in Eq. (2) to calculate C_w.

Sample concentration as a function of pump and fracture depths

Flushing and purging the water lines in a well-pump system prior to collecting a sample is common practice. To examine how purging or pumping time can influence the flow-weighted averaging effect on water from different fractures, advection and dispersion calculations were performed in Microsoft Excel. The transport time of water was calculated from each of two fractures to a downhole pump. The fractures are each assigned a transmissivity and contamination level. The model also examines how the depth of the pump in relation to the depth of the fractures can affect the total concentration at the pump through time. The model uses the 1-D Ogata and Banks equation for advection and dispersion (Ogata and Banks 1961) and applies it to each fracture in the well:

$$C_p = \frac{C_0}{2} * \left[erfc\left\{ \frac{x - vt}{2\sqrt{Dt}} \right\} + e^{\frac{vx}{D}} * erfc\left\{ \frac{x + vt}{2\sqrt{Dt}} \right\} \right] \tag{4}$$

where C_p is the concentration observed at the pump from a single fracture at time t; C_0 is the concentration in a contributing fracture; x is the distance between a fracture and the pump; v is the velocity $= \frac{\text{Fracture discharge}}{\text{Cross sectional area of wellbore}}$; t is the time; D is the Taylor dispersion coefficient. It is assumed that dispersion in the well while pumping follows the Taylor dispersion for flow in a pipe (Taylor 1954). A very low value for D was used (0.01 m^2/s) as a conservative calculation. A larger value of D would take a much longer time for concentrations to reach the maximum value at the pump.

The average concentration at the pump (C_{pt}) is determined by first applying Eq. (4) to each fracture, which calculates the concentration at the pump within an elemental volume of water from each fracture at a time t (C_p). Then, an average concentration (C_{pt}) weighted by discharge is calculated using Eq. (2).The model assumes that C_w starts at zero, pumping discharge remains constant from $t=0$, and discharge from each fracture into the well while pumping remains constant.

Monitoring drawdown in private drinking wells during household usage

The above assessments provide insight into how water levels in the well can affect concentrations in the well. To evaluate how much drawdown might take place in domestic wells, downhole pressure transducers were installed in three private domestic drinking wells in the Hebron Gneiss (a fractured granitic gneiss) in Storrs, Connecticut, USA, over a period of three weeks to monitor typical drawdown ranges under normal household use. Pressure measurements were recorded every 15 min. The data were used to calculate the daily drawdown range in three wells.

Pumping tests to evaluate changes in concentration as a function of purging duration

Pumping tests were conducted in three bedrock wells to evaluate how concentrations change with pumping duration. Each well had a history of elevated chloride levels near or above the EPA Secondary Maximum Contaminant Level (SMCL) for drinking water of 250 ppm. Two of the wells, SC-1 and TH-1, are drilled into marble and gneiss in Sherman, CT. The wells are of typical domestic well construction with a diameter of 15.24 cm, steel casing from the surface into 3 m of rock, and are open boreholes to depths of 30.5 m and 91.5 m, respectively. The third well, Sima-1, is drilled into granitic gneiss in Storrs, CT and constructed in a similar manner to the other two wells to a total depth of 89.9 m. The pumping tests in SC-1 and TH-1 were conducted through un-filtered faucets connected to the well pump systems. Sima-1 is an open bedrock well with no dedicated pumping system. For this pumping test, a 370-watt domestic well pump was lowered to a depth of approximately 14 m above the bottom of the well to simulate a typical domestic well configuration. The well was pumped at a constant rate for five hours. To evaluate how the water quality in the wellbore changed with pumping, a YSI 6600 downhole sonde was used to develop specific conductivity depth profiles before and after the pumping test. Profiles obtained were compared to the depths of contributing fractures and borehole flow conditions as determined previously with heat pulse flowmeter and geophysical logging (Cagle 2005).

Throughout the duration of each of the three pumping tests, Hach Chloride Test Strips were used to measure chloride levels. Based on calibrating test strip concentration measurements with standard laboratory analyses, the test strips were found to have an accuracy of better than 5%. The discharge was measured through two flow meters, one that measured flow rate, and another that measured purge volume. At least three wellbore volumes were pumped from each well.

Results

Model results

Concentration as a function of drawdown

The concentration with drawdown model results are shown in Fig. 3. Table 1 lists the parameters used in the model calculations. All calculations used constant values for well diameter (0.152 m) and storativity (5.00E−05). The values for fracture transmissivity, head, depth and total well depth used in this modeling calculation were based on USGS Test Well 103R in Storrs, Connecticut, USA, in the Hebron Gneiss formation (Johnson et al. 2002). In these model cases, $h_1 > h_2$ so it is assumed that $C_w = C_1$ under static conditions. Figure 3a shows that the ratio of Q_2/Q_1 increases with drawdown until it reaches the ratio of T_2/T_1. Figure 3b shows that the concentration in the well increases as a function of drawdown for the case where the lower head fracture is contaminated. Figure 3c shows the opposite, where the concentration decreases as a function of drawdown for the case where the higher head fracture is contaminated.

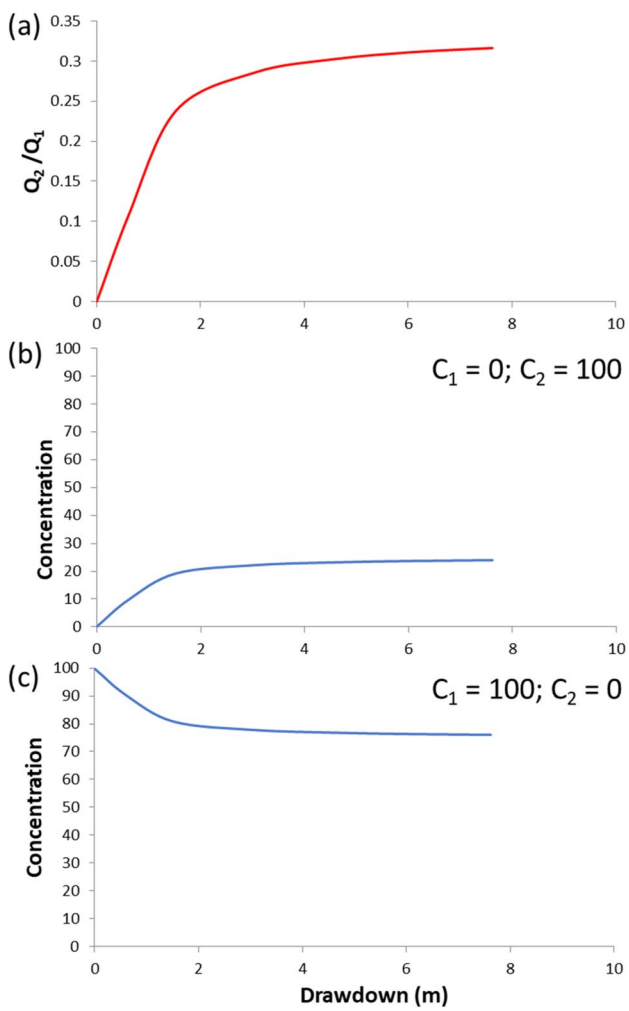

Fig. 3 Results of the concentration with drawdown calculations where $h_1 > h_2$. **a** Shows how the ratio of fracture Qs changes as a function of drawdown. Curves **b** and **c** show how the average concentration in the wellbore changes depending on which fracture is contaminated

Table 1 Hydrogeologic parameters used in Fig. 3 calculations of concentration as a function of drawdown

Parameter	Symbol	Unit of measure	Measured value
Head in open well	H_w	(m)	171.47
Head in fracture 1	H_1	(m)	171.57
Head in fracture 2	H_2	(m)	171.36
Transmissivity in fracture 1	T_1	($m^2\ s^{-1}$)	2.26E−5
Transmissivity in fracture 2	T_2	($m^2\ s^{-1}$)	7.42E−6
Depth of fracture 1	d_{F1}	(m)	9.76
Depth of fracture 2	d_{F2}	(m)	25
Depth of well bottom	d_w	(m)	33.5

Sample concentration as a function of pump to fracture distance

Figure 4 shows the results for the sample concentration as a function of pump to fracture distance calculations. Table 2 contains the parameters used throughout the six modeled cases. Individual fracture transmissivity ratios and pump depths were changed between different cases and their values are noted in Fig. 4. F_1, the shallower fracture, has a concentration of 0 and F_2 has a concentration of 100. Table 3 lists the contaminant arrival times at the pump for each case in Fig. 4. Table 3 also summarizes how the maximum sample concentration at the pump is impacted by changes in individual fracture transmissivity. As would be expected, the advective front and maximum sample concentration arrival times increase as the distance between the pump and contaminated fracture is increased and as the transmissivity of the contaminated fracture is reduced. The maximum concentration achieved is determined by the ratio of transmissivities between the two fractures.

In-well testing results

Drawdown exhibited in Storrs, CT domestic wells

Figure 5 shows the daily drawdown range exhibited in three domestic bedrock wells. During the three-week test period, significant variations were observed in the water levels in the wells. On one or more occurrences per day the three wells exhibited drawdowns of at least 0.59 m, 0.89 m, and 1.14 m.

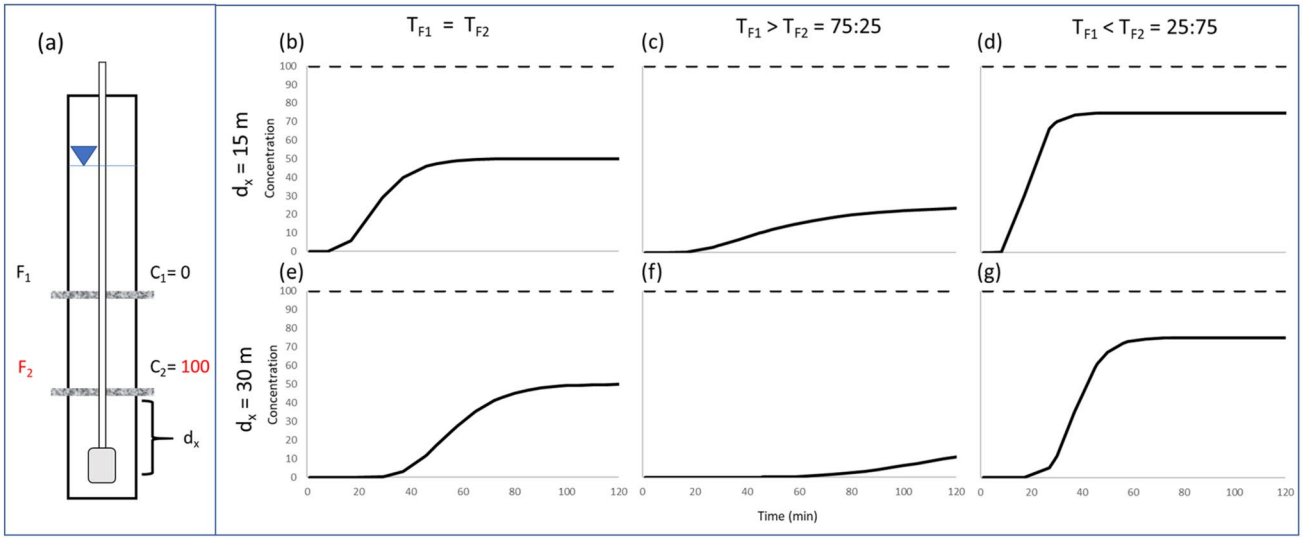

Fig. 4 Results of the calculations of concentration as a function of pump to fracture distance. **a** A diagram of the two fracture well used in the calculations **b**–**g** show changes in concentration at the pump as a function of time while pumping continuously. Fracture transmissivity and pump depth are varied between each case while all other parameters remain constant

Table 2 Hydrologic well parameters used in Fig. 4 calculations of concentration as a function of pump to fracture distance

Parameter	Symbol	Unit of measure	Value
Well diameter	–	(cm)	15.24[a]
Depth of fracture 1	d_{F1}	(m)	10.9
Depth of fracture 2	d_{F2}	(m)	40
Depth of pump	d_p	(m)	55 and 70
Distance of pump from F_2	dx_2	(m)	15
Concentration of fracture 1	C_1	(%)	0
Concentration in fracture 2	C_2	(%)	100
Discharge of pump	Q_p	(L min^{-1})	18.9[a]
Dispersion coefficient	D	(m^2 s^{-1})	0.01

[a]Values chosen based on typical domestic well construction

Table 3 Advective front arrival time, maximum concentration arrival time, and maximum concentration achieved at the pump in Fig. 4 calculations

Case	Advective front arrival time (min)	Max concentration arrival time (min)	Maximum concentration achieved
b	27	> 60	50
c	51	> 130	25
d	18	> 27	75
e	56	> 100	50
f	127	> 300	25
g	38	> 65	75

On one or more occurrences per week, the three wells exhibited drawdowns of at least 1.33 m, 2.25 m, and 3.53 m. The maximum drawdown exhibited in a well throughout the testing period was 4.35 m.

Effects of pumping duration and purge volume on chloride concentrations

Figure 6 shows the chloride concentration results throughout the duration of each pumping test in the wells SC-1, TH-1, and Sima-1. Table 4 contains the parameters of each pumping test including pump discharge, maximum drawdown (when measured), well depth, depth to water, pump depth, and borehole volume.

During the pumping tests, each of the three wells exhibited different concentration changes. SC-1 (Fig. 6a) exhibited minimal variation in chloride concentration while pumping three borehole volumes over three hours, ranging from 196 to 216 ppm. TH-1 (Fig. 6b) had an initial chloride concentration of 162 ppm at the spigot. The concentration increased to 256 ppm after 2.64 borehole volumes were pumped from the well over 198 min, a level which exceeds the 250 ppm EPA secondary maximum contaminant level (SMCL) for drinking water quality. Sima-1 (Fig. 6c) had an initial chloride concentration at the pump outlet of 285 ppm. The chloride concentration increased to a maximum of 310 ppm after 0.70 borehole volumes were pumped from the well over 63 min. After 224 min and 2.41 borehole volumes were pumped, the concentration fell to 238 ppm, below the SMCL of 250 ppm. After 284 min and

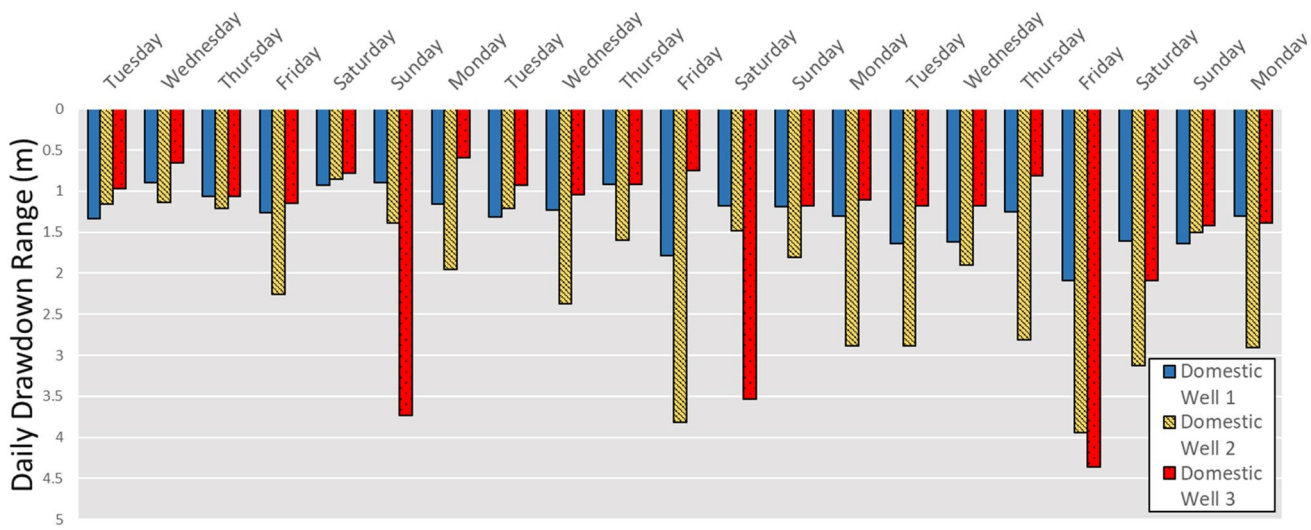

Fig. 5 Normalized downhole pressure transducer data showing the daily range of drawdown exhibited by three domestic bedrock drinking water wells in Storrs, CT, USA

3.1 borehole volumes were pumped, the concentration fell to 196 ppm, its minimum value throughout the test.

Figure 7 shows the specific conductivity as a function of depth in Sima-1 before and after the pumping test was conducted. Under ambient conditions prior to pumping, the specific conductivity was 1212 µS/cm at the water table and gradually increased to 1249 µS/cm at the bottom of the well. Prior downhole geophysical characterization (Cagle 2005) indicates that there are three primary contributing fractures in this well at 16.5 m, 39.3–41.8 m, and 85.9 m below top of casing (BTOC). Fracture hydraulic head measurements (Flahive 2017) indicate that there is a 0.8 m downward gradient between the upper and lower fractures. Prior flow profiles (Vitale and Robbins2017) indicate that the upper two fractures flow into the well under ambient conditions, and majority of the well outflow occurs via the deepest fracture at the 85.9 m BTOC. The post-pumping depth profile was conducted immediately after the pumping test concluded. The depth of the pump was 76 m BTOC, and 3.5 borehole volumes were pumped from the well over 314 min. The specific conductivity showed variation throughout the well with levels ranging from 1336 µS/cm near the top of the water column, to 819 µS/cm at 89.9 m BTOC, the bottom of the well. The specific conductivity profiles indicate that the fracture at the 85.9 m BTOC, which is outflowing under ambient conditions, contains water with lower conductivity than the upper fractures and that flow was reversed at some point during the pumping test. As pumping continued, the water from this fracture flowed toward the pump inlet at 75 m BTOC and reduced the conductivity of the pump discharge water.

Discussion

As noted above, when domestic well water samples are collected, drawdown, discharge rate, and pump depth are not typically monitored or recorded. As shown by the modeling results and further supported by the pumping tests, these are important factors that influence the flow-weighted average concentrations obtained when sampling a fractured rock well. The drawdown alone determines whether a fracture will or will not contribute water to the well depending on the degree of drawdown and the hydraulic head in each fracture. Furthermore, the fracture transmissivities and the drawdown determine the relative contributions of water from each fracture. Despite the importance of drawdown in the well during sampling, as noted, it is typically not measured during domestic well water sample collection. Furthermore, the variable drawdown measurements observed in the three domestic wells as a function of time under normal household usage conditions (Fig. 5) could result in changes in water quality during usage, as suggested by the modeling and confirmed by the pumping tests.

The results in Fig. 6 demonstrate how the water quality in fractured bedrock wells can vary significantly with extended pumping, even when wells are drilled into the same formation, as is the case with SC-1 and TH-1. Each of the three pumping test wells had different total depths, hydraulic heads, drawdown achieved during pumping, and most likely, different fracture distributions, which further demonstrates the variations found among typical domestic wells in fractured rock. SC-1, the shallowest wells in

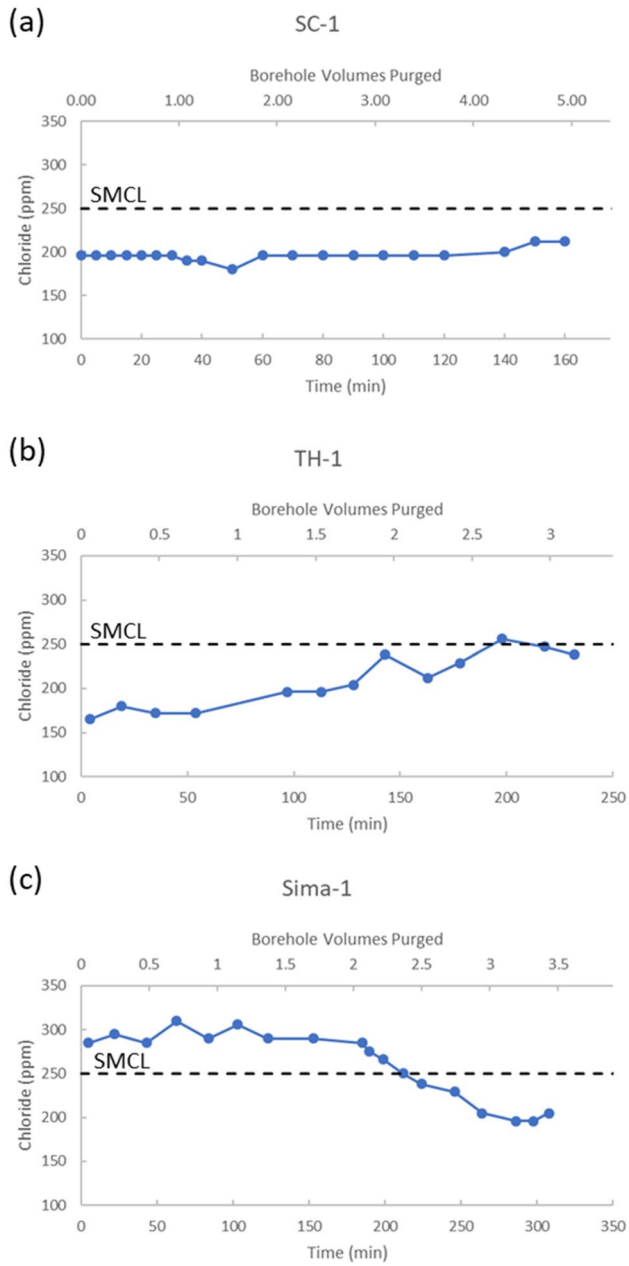

Fig. 6 Chloride concentration measurements vs. time and borehole volume purged in wells **a** SC-1, **b** TH-1, and **c** Sima-1. The dashed line indicates the EPA's secondary maximum contaminant level (SMCL) for chloride

this study, exhibited no change in concentration during the test, which indicates the likelihood of a dominant fracture zone with a much greater transmissivity than the others, and/or that all contributing fractures contained similar chloride concentrations. TH-1 exhibited a gradual rise in concentration over 198 min and 2.64 borehole volumes of purging. This indicates a likelihood of a low head fracture that contains a higher chloride concentration than the fracture(s) that contribute to the well under non-pumping

Table 4 Pumping tests parameters for three bedrock wells in which chloride concentrations were monitored

Parameter	Unit of measure	Well		
		SC-1	TH-1	Sima-1
Well diameter	(cm)	15.24	15.24	15.24
Borehole depth	(m)	30.5	91.5	89.9
Initial depth to water	(m)	5.03	15.3[a]	2.97
Depth of pump	(m)	24.4	70.1	76.0
Discharge of pump	(L/min)	15.12	18.9	18.9
Maximum drawdown	(m)	2.56	–[b]	9.47
Borehole volume	(L)	473	1416	1617

[a]Approximate depth to water based on historic data

[b]Downhole measurements while testing were not available in this well

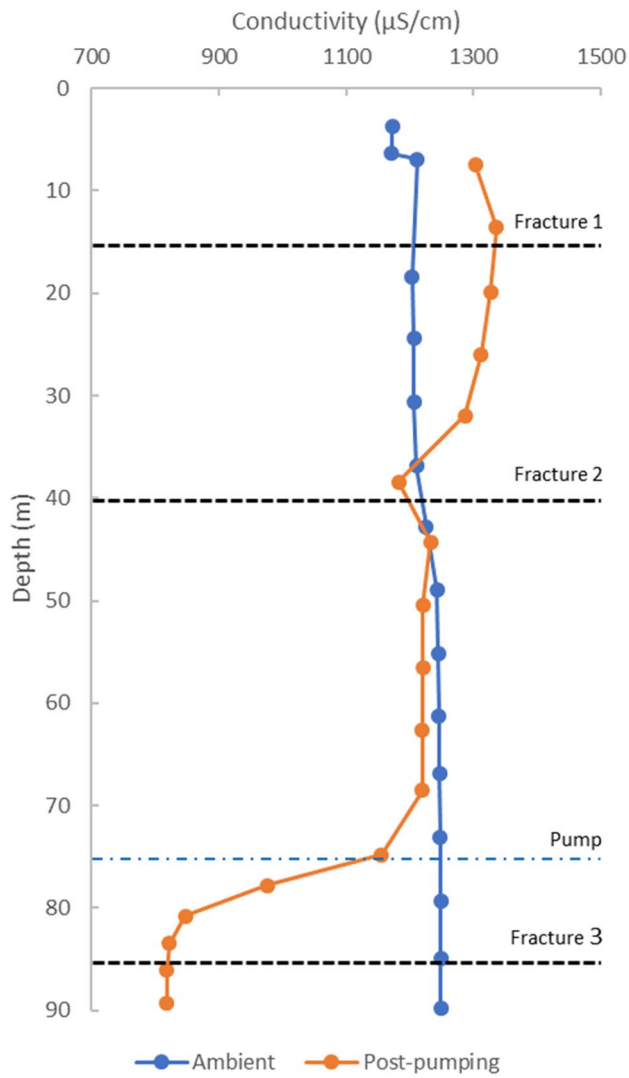

Fig. 7 Specific conductivity as a function of depth during ambient conditions and post-pumping conditions in well Sima-1

conditions. The relative transmissivity of this fracture could be been lower, and/or it could be distant from the pump inlet depth, resulting in the gradual 36% increase in chloride concentration observed throughout the test. Sima-1, which exhibited a relatively stable chloride concentration in the first 3 h and 2 borehole volumes of purging, declined in concentration by 30% while purging continued for another 2 h and 1.5 borehole volumes. This is attributed to a fracture with lower chloride concentration and head existing in the well. This fracture, which is typically outflowing under non-pumping conditions, began flowing into the well towards the pump after significant drawdown was achieved. This is likely due to this fracture having a lower transmissivity than cumulative transmissivities of the two higher concentration fractures, which would result in a slow travel time between this fracture and the pump.

The two downhole specific conductivity profiles conducted in Sima-1 show a significant difference in water quality under ambient versus post-pumping conditions and provide more insight as to how flow within the wellbore changed during pumping (Fig. 7). With downward flow throughout the well under ambient conditions, the specific conductivity in the entire water column is relatively uniform and represents an average of the specific conductivities in the upper two inflowing fractures. The post-pumping profile shows a distinctly different specific conductivity distribution throughout the well, with sharp decreases near the depths of each fracture and the pump inlet. This highlights that each of the three fractures contains a different water quality. After pumping, the conductivity was higher than ambient levels between the water table and fracture 2 (5–40 m), owing to fracture 1 (16 m) containing the highest conductivity and flowing downward during pumping. The conductivity was similar to ambient levels between fracture 2 and the pump (40–75 m), as this consists of a mixture of water from fractures 1 and 2, which were both flowing down toward the pump. The conductivity is lower than ambient beneath the pump, as fracture 3 contains the lowest specific conductivity, is typically flowing out of the well during ambient conditions, and began flowing into the well towards the pump during the pumping test. The results in Fig. 7 demonstrate how water quality can vary greatly as a function of depth and pumping conditions.

The models and test results in this study provide insight to improve sampling for contaminants in domestic wells by demonstrating the influence that many of the discussed factors can potentially have on the water quality of a collected sample. Under an ideal situation, the hydrogeologic conditions of these wells would be fully characterized; however, this would be impractical for domestic wells given the costs for such characterization. Given a data objective to determine whether contaminant concentrations in a domestic well exceed an MCL, when the well is initially sampled, it should be pumped continuously for an extended period and multiple water samples should be collected periodically to determine how the concentration changes with pumping duration. Based on trends observed in the concentration throughout the pumping duration, an optimal purging time specific to a particular well can be established for future sampling that achieves a more representative, or at least a consistent, concentration for risk assessment. If concentrations are observed to rise with purging time, as in Fig. 6a, b longer purging duration would be recommended for future sampling events. If concentrations remain the same or fall with purging duration as in Fig. 6a, c, a short purging duration would be recommended for future sampling, sufficient to fully flush the system piping. It is also recommended that the purging time be recorded along with any future samples that are collected.

The factors as described above also influence evaluating the degree of correlation between natural contaminant concentrations determined in domestic wells and the lithogeochemistry of the underlying rock. As such, caution should be taken when using domestic well data to ascribe contaminant risk factors with geologic formations in the absence of other confirmatory datasets, e.g., rock geochemical analyses.

Conclusions

The results of this study demonstrate that water samples collected from domestic wells in fractured crystalline bedrock, where fractures have different hydraulic heads and water qualities, represent weighted average concentrations. As such, the factors that influence concentration averaging should be considered in developing protocols for sample collection and interpreting concentration data from domestic wells. Relying on water quality data from domestic wells without considering the factors discussed in this study can lead to highly misleading interpretations with respect to such assessments as identifying groundwater contaminant sources and mapping contaminant distributions. Recommendations are made in this paper to improve sample collection; however, given the influence that drawdown and pumping duration may have on water quality, it must be realized that to obtain more representative samples of what people are drinking, you must sample water at different periods of time while they are using their water. Furthermore, to obtain samples representative of fracture conditions to conduct contaminant mapping requires borehole logging and the use of discrete multi-level sampling methods.

Author contributions MAH: Conceptualization, Methodology, Investigation, Writing—original draft, Writing—review and editing, Visualization, Project administration. GAR: Conceptualization,

Methodology, Investigation, Resources, Writing—review and editing, Supervision, Funding acquisition. MJM: Conceptualization, Methodology, Investigation.

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