Abstract
Gravity-driven infiltration into the shallow subsurface via small-diameter wells (SDWs), i.e., wells with an inner diameter smaller than 7.5 cm (3 inches) and no gravel pack) has proven to be a cost-efficient and flexible tool for managed aquifer recharge (MAR), as it provides relatively high recharge rates with minimal construction effort. SDWs have a significantly smaller open filter area than larger diameter wells with gravel pack, making the infiltration of low-quality waters through these wells more at risk clogging. To investigate their susceptibility for biological and physical clogging, 24 physical models with different well setups were evaluated by infiltrating either nutrient-poor but turbid water or nutrient-rich but clear water. The experiments showed that smaller diameters and the lack of a gravel pack increase the well’s susceptibility to both kinds of clogging. However, this effect was observed to be much more pronounced for physical than for biological clogging. Our conclusion is that SDWs show severe disadvantages with respect to the infiltration of highly turbid waters in comparison to large diameter wells with a gravel pack. Nevertheless, this disadvantage is much less severe when it comes to the infiltration of clear but nutrient-rich waters (e.g., treated wastewater). Depending on the economic and geological circumstances of a MAR-project, this disadvantage could be outweighed by the significantly lower construction costs of SDWs.

Introduction
Managed aquifer recharge (MAR) is becoming more popular in recent years as one option to combat water scarcity. Transient water shortages (e.g., during the dry seasons in summer time) can be buffered efficiently at reasonable costs if geological conditions are feasible. Furthermore, percolation via the vadose zone can improve the quality of the recharge water significantly (Goren et al. 2014) which is used during soil-aquifer treatment for the in situ treatment of pretreated waste water (e.g., Sharma and Kennedy 2017; Fichtner et al. 2019). Finally, MAR can be used for other purposes in groundwater hydraulics, for example, to combat saltwater intrusion (Ebeling et al. 2019).

Bouwer (2002) and Sprenger et al. (2017) discussed common systems for infiltration of surplus waters such as basins, trenches and wells. Well infiltration offers several advantages, as it allows targeting well-permeable sandy layers for infiltration, whereas during basin infiltration the water passes the whole soil profile. For the latter, pronounced heterogeneous structures and impermeable layers can inhibit infiltration substantially (Mawer et al. 2016; Pyne 2005; Sprenger et al. 2017). Small-diameter wells (SDW) have recently been the subject of both numerical (Händel et al. 2014) and experimental studies (Händel et al. 2016; Liu et al. 2016). SDWs are usually smaller than 3 inches (inner diameter <7.5 cm) and do not have a gravel pack, which brings the advantage of a quick, easy, and very cost-efficient construction (Händel et al. 2016). Shallow SDWs with a depth of 5–10 m can be installed, for instance, with percussion drill or a mobile direct-push rig in a very short time, often within hours (Dietrich and Leven 2009).
The referenced studies investigated only the potential of SDW for clean water recharge. In practice, however, storm water (Page et al. 2010; Gao et al. 2014), water from ephemeral rivers (Alataway and El Alfy 2019), and treated waste water (Goren et al. 2014; Sun et al. 2020) are used for MAR. While storm waters contain high loads of suspended particles, treated waste waters are usually characterized by high concentrations of nutrients, bacteria, and dissolved organic matter (Icekson-Tal et al. 2003; Asano and Cotruvo 2004). Surface waters can contain both nutrients and suspended solids (Casanova et al. 2016; Sprenger et al. 2017). Depending on the infiltration water’s origin, clogging during infiltration by physical, chemical, and/or biological processes can lead to significant reductions in recharge capacities (Olsthoorn 1982; Pyne 2005; Jeong et al. 2018).

Infiltration of highly turbid water may result in physical clogging, that is, the blocking of well screen openings and pore channels within the porous medium (Olsthoorn 1982). High nutrient loads may cause microbial growth which leads to biological clogging. Here, the biomass, that is, the microorganisms themselves and their secreted biofilm consisting of extracellular polymeric substances can block both pore channels and the screen openings of wells (Vandevivere and Baveye 1992; Camprovin et al. 2017). An exhaustive review on the different clogging types is given in Jeong et al. (2018).

To limit the overall effect of clogging on MAR operations, Bouwer (2002) recommended surface-infiltration systems such as infiltration basins or, alternatively, active pretreatment before injection of these clogging-associated waters into wells, for example, via screens and filters to remove debris (Jeong et al. 2018). Basins are simple to clean, for example, by plowing or excavation of the uppermost layer, but need comparatively large areas and are associated with other potential problems such as higher evaporative losses or mosquito breeding (Jeong et al. 2018). Here, large-diameter aquifer-storage-and-recovery wells, equipped with a gravel pack, are a common and well-established alternative (Pyne 2005). These wells are considered to be less susceptible to clogging than the comparably much cheaper SDW’s, which is to be studied here.

In sandy aquifers, physical clogging by clay to silt-sized turbids mainly affects the immediate well-aquifer interface as the particles mostly block the pores in the first centimeters of the porous media (Barquero et al. 2019). With regard to biological clogging, Mostafa and Van Geel (2007) conducted laboratory column experiments revealing that biological clogging is also restricted to the first few centimeters from the entering point of the nutrient-rich water. Furthermore, they observed that smaller pores are significantly more affected than larger ones. When transferring these one-dimensional experiments to recharge wells, it is expected that both biological and physical clogging will mainly take place at the well-aquifer interface, either directly at open slots of the well screen being in contact with the aquifer or between gravel pack and aquifer (see Figure 1). The last assumption might seem oversimplified, because the interface between well screen and gravel pack can also be prone to accumulation of particles and microbial growth. However, it is considered to have a much smaller effect here as the wider screen openings and larger pores provide more buffer for biomass/particles. This assumption is also supported by the findings of Mostafa and Van Geel (2007).

Decreasing the active filter area will offer less space for particle and biofilm accumulation. SDWs, typically without a gravel pack, would thus be more susceptible for clogging of any kind. However, as the actual effect has not yet been explicitly investigated, neither quantitatively nor qualitatively, this conclusion is solely based on intuition.

Physical clogging can be described as a continuous accumulation of particles on the interface area, meaning that an increase in open filter area will have an inversely proportional effect on clogging rate. The case is far more complex for biological clogging, where different effects come into play. On the one hand, a smaller interface area is usually accompanied by an increase in flow velocity. This can impact the superficial morphology of the biofilm which in turn may consequently lead to an earlier decrease in hydraulic conductivity (e.g., Kim et al. 2010). On the other hand, Thompson et al. (2015) correlated higher infiltration rates with higher shear stresses and, hence, a lower clogging rate. And finally, biofilm density and clogging rate are directly influenced by the composition of the infiltrated water and its nutrient concentration. These effects are not affected by the well setup (Kim et al. 2010). This raises the suspicion that clogging rates are less affected by well diameter and gravel pack during biological clogging than during physical clogging.
Within the present study, physical and biological clogging is experimentally assessed by employing a variety of different well-aquifer model setups with varying well diameters and gravel packs in order to derive conclusions on the impact on clogging rate. For this purpose, an evaluation approach is developed based on an inversely linear model (physical clogging) and an adapted four-parameter logistic (4PL) model (biological clogging). Conclusions are drawn on SDW clogging and to what extent it might be an obstacle for low-quality water infiltration.

Materials and Methods

Theoretical Background (Open Filter Area)

The open filter area ($A_{\text{Filter}}$) is the area between either the well screen openings and the aquifer or the gravel-pack’s effective porosity and the aquifer. It is the area where water can actually flow and where (microbial or mineral) mass accumulation has significant hydraulic impact. A straightforward approach to quantify $A_{\text{Filter}}$ is

$$A_{\text{Filter}} = A_{\text{Well}} * n_s = 2 * \pi * r_w * b_{\text{well}} * n_s \quad (1)$$

for a well with a given screen length ($b_{\text{well}}$), effective diameter ($r_w$), and relative open screen area ($n_s$). In this approach, $A_{\text{Filter}}$ will directly correlate with $r_w$ and $n_s$. For an SDW, $n_s$ will primarily depend on slot size and density. If a gravel pack is present then the direct connection between the natural aquifer material and the well screen is, however, cut. In these scenarios our simplified but not simplistic approach assumes that $A_{\text{Filter}}$ (and $r_w$) is effectively located at the interface between the gravel pack and the aquifer and that $n_s$ can be approximated by the gravel pack’s effective porosity (see Figure 1).

Experimental Procedure: Physical and Biological Clogging

Two experimental series A and B (as schematically presented in Figure 3), each with six well-aquifer models, were set up to conduct tests for both physical and biological clogging. In total, four experimental runs were conducted: $A_{\text{Phy}}, B_{\text{Phy}}, A_{\text{Bio}},$ and $B_{\text{Bio}}$. Hereby, the subscripts refer to the clogging type. Apart from the wells in the gravel aquifer, each well model was set up with a physical replicate (identical setup).

In the experimental series $A_{\text{Phy}}$ and $A_{\text{Bio}}$, SDWs without gravel pack (screen diameters of 63 mm and 32 mm) were compared to each other and with rather “conventional” wells of the same screen diameters (63 mm and 32 mm) but equipped with an gravel pack (125 mm diameter). The absolute open-filter areas between the two SDW variants were considered to differ by a factor of two as only the diameter changed. In contrast, the conventional wells produced both a doubling in effective diameter and a significant increase in the open filter area (effective porosity of the gravel pack $[0.3 \pm 0.03]$ vs. relative open screen area of the pipes $[0.05]$). For the conventional wells with 63 mm and 32 mm, $A_{\text{Filter}}$ was estimated to be 12 to 24 times larger than for their respective SDW counterparts.

The experimental series $B_{\text{Phy}}$ and $B_{\text{Bio}}$ compared a standard SDW (screen diameter: 63 mm) with a smaller (screen diameter: 32 mm) conventionally built well of the same effective outer diameter (63 mm including gravel pack) thus increasing $A_{\text{Filter}}$ by a factor of roughly four. Additionally, wells of both 32 mm and 63 mm screen diameters were installed directly in a gravel aquifer to assess clogging within the gravel pack material itself. Setups for both experiments are shown in Figure 3.
Figure 2. Experimental setup. (a) Basic concept of a single well-aquifer model with (right) and without gravel pack (left). Pressure head difference between the well and the drainage is defined by equation $h_w - h_o$. Variables $r_W$ and $r_o$ are the radii of the well and the model. $H$ is the thickness of the sediment and $Q$ is the injection flow rate. (b) Experimental setup for the clogging tests. Water was pumped from a reservoir to a free overflow, from where it was delivered to the wells with a constant flow rate. The flow channels were adjusted to provide the same rate for all well-aquifer models. Excess water (dashed arrows) was either channeled back to the reservoir (biological clogging experiments) or discarded (physical clogging experiments).

Figure 3. Two models were set up for each scenario in the experimental series A and B where well radius ($r_W$) and gravel pack were varied to assess the difference between physical ($A_{Phy}$ and $B_{Phy}$) and biological clogging ($A_{Bio}$ and $B_{Bio}$).

The infiltration rate was set to values between 3 and 3.5 mL/s via a fixed overflow. For all experimental series, the aquifer-well-models were first rinsed with clear and degassed tap-water for 48 h. After that, that is, at experimental time 0:00 the clear water was replaced by either a clay suspension or a nutrient solution.

For the biological clogging experiments ($A_{Bio}$ and $B_{Bio}$), a circular system (see Figure 2b) was established: specifically, tap water was pumped in a closed loop and nutrients were constantly added with a peristaltic pump and a nutrient solution (30 mg/L/d dissolved organic carbon [DOC]; 5 mg/L/d NH$_4^+$). The resulting nutrients in the circulating water were meant to be similar to literature values for treated wastewater (Ben Moshe et al. 2020). A DOC amount of 30 mg/L was added (“on top”) in the very beginning of the experiment to give a starting boost for the microbial activity. During the physical clogging experiments ($A_{Phy}$ and $B_{Phy}$), the water was not circulated in a closed loop, but was directly taken...
from a continuously stirred reservoir with an initially defined concentration of 200 mg/L of a clay–silt–sand mix (43% clay, 44% silt, and 13% sand) and all outflow from the models was discarded.

**Data Evaluation Concept: Assessment of Clogging Using Dupuit-Thiem**

The clogging intensity was indirectly quantified by observing the pressure head difference \((h_w - h_o)\) between the well and the drainage. This difference was used to derive the change in hydraulic conductivity \(K\) by applying the Dupuit-Thiem equation after Hölting and Coldewey (2013) and with symbology as described in Figure 2, that is,

\[
K = \frac{[\ln(r_o) - \ln(r_w)] \ast Q}{2 \pi \ast H \ast (h_w - h_o)} \tag{2}
\]

The derived hydraulic conductivity value reflects the hydraulic properties of the aquifer and the well itself as it is directly affected by any ongoing clogging. A correction for turbulent flow in the near vicinity of the well (Forchheimer 1901; Bear 1972) seems adequate. However, this was not possible in the scope of this study due to the clogging-induced anisotropy. The required hydraulic head difference and the flow rate were monitored manually for the experiments on physical clogging, which lasted not longer than 6h. As the experiments on biological clogging lasted longer (3–5 days) the hydraulic head was monitored each minute by employing pressure sensors (Van Essen Instruments, Mini-Diver DI501) and a moving average was calculated for a subset of five measurements (15-min interval) to minimize measurement noise caused by turbulences in the water column. Flow rates were measured manually several times a day and corrected if necessary by linear interpolation of the pumping rate.

**Data Evaluation: Approaches for Physical and Biological Clogging**

The most obvious way to compare the different setups is by a comparison of the time, when a certain state of clogging is reached, for example, when \(K\) drops to 10% of the initial \(K_{max}\). However, the dynamics of the clogging processes may differ strongly. Biological clogging, for instance, starts only after a certain period of time given that a nutrition supply exists. Contrary, physical clogging will commence almost immediately at the moment when the first turbids are reaching the well screen. Thus, this time comparison cannot be the only evaluation factor. For further comparison, an empirical approach was tested for both physical and biological clogging. While assuming that flow rate and turbidity remain constant, Pyne (2005) suggests a linear relationship between time \(t\) and hydraulic resistance \(R(t)\) for describing physical clogging, that is,

\[
R(t) = m \ast t + R_0, \tag{3}
\]

where \(R_0\) defines the hydraulic resistance before the start of the physical clogging process. Here, \(R_0\) equals the reciprocal of the maximum hydraulic conductivity \(K_{max}\). \(m\) is the clogging velocity which mainly depends on turbidity, flow velocity and soil texture. In order to relate this to hydraulic conductivity measured after the final state of physical clogging \(K_p\), the reciprocal of Equation 3 was taken as shown in Equation 4.

\[
K_p(t) = \frac{1}{R(t)} = \frac{1}{m \ast t + R_0} = \frac{1}{m \ast t + \frac{1}{K_{max}}} \tag{4}
\]

Conversely, there is no general relationship between time and loss in hydraulic conductivity for biological clogging scenarios, as the temporal behavior here is more complex and depends on a multitude of experimental parameters such as nutrient load, temperature, and sediment texture. In this study the microbial growth is assumed to approximately follow Monod kinetics (Monod 1949). It is assumed that biomass will slowly but inexorably fill the available pore space. This will, however, only affect hydraulic conductivity when the amount of biomass exceeds a certain level and begins to partially block the main water flow pathways (Mostafa and Van Geel 2007). After reaching this point, hydraulic conductivity will start to drop faster until the available pore space starts to become the limiting factor. The drop in hydraulic conductivity will then decelerate and finally plateau at a certain level. This development can be explained by a 4PL model, also called Hill model (Robertson 1908), which is often used to describe biological processes (Motulsky and Christopoulos 2003). The model uses four parameters: two to determine the start and final value of the measured size and two to determine the shape of the curve (delay and decrease velocity). With regard to the latter two, the equation shows a slight weakness when clogging delay and clogging velocity are evaluated separately, as the value for delay also influences the steepness of the curve. To isolate effects on steepness to one parameter only, the nature of the Hill model was imitated with a logistic function as shown in Equation 5.

\[
K_b(t) = K_{min} + \frac{K_{max} - K_{min}}{1 + e^{b(t-c)}}, \tag{5}
\]

where \(K_b\) is the hydraulic conductivity for biological clogging, \(K_{max}\) and \(K_{min}\) are the hydraulic conductivities at the beginning and the end of the experiment, \(c\) is the inflection point of the curve (here: the time when the relative hydraulic conductivity drops to 50%) and \(b\) is the steepness of the decline.

Comparing the two clogging forms with each other in order to evaluate whether physical or biological clogging is affected more by the well setup, seems rather difficult. Nevertheless, in order to quantitatively evaluate differences in clogging dynamics, the steepness factors \(m\) (see Equation 4) and \(b\) (see Equation 5) were compared by assessing their impact on the highest gradient on the curve. For the “physical clogging” function, this steepest point is to be found at the very beginning of the
experiment, and the highest gradient equals to the first derivative \( K_p' \) at \( t = 0 \), that is,

\[
K_p'(t) = -\frac{m}{(m \cdot t + K_{\text{max}}^{-1})^2} \Rightarrow K_p'(0) = -m \cdot K_{\text{max}}^2.
\]  

(6)

Equation 6 indicates a linear relationship between the steepness \( K_p' \) at this time and the steepness factor \( m \). As \( K_{\text{max}} \) represents the initial hydraulic conductivity, it can be considered as constant and measurable. For the “biological clogging” function, the point with the highest steepness is the curve’s inflection point \( c \) which is determined by calculating the first derivative \( K_b' \) at \( t = c \), that is,

\[
K_b'(t) = \frac{(K_{\text{max}} - K_{\text{min}}) \cdot b \cdot e^{b(t-c)}}{(1 + e^{b(t-c)})^2} \Rightarrow K_b'(c)
\]

\[
= -\frac{b}{4} \cdot (K_{\text{max}} - K_{\text{min}}).
\]

(7)

While the steepness, again with a linear relationship to the steepness factor \( b \), might also depend on the finally reached minimum hydraulic conductivity \( K_{\text{min}} \) the latter is usually one to two magnitudes smaller than \( K_{\text{max}} \). Hence, its effect on the can be considered negligible.

To eventually compare the two steepness factors, the derivations at the curve’s steepest points can be set equal, that is, \( K_p'(c) = K_b'(0) \). This allows for transforming the “biological clogging factor” \( b \) into an “equivalent physical clogging factor” \( m_{\text{eq}} \), that is,

\[
m_{\text{eq}} = \frac{b}{4 \cdot K_{\text{max}}} \cdot (K_{\text{max}} - K_{\text{min}}) \approx \frac{b}{4 \cdot K_{\text{max}}}.
\]  

(8)

The latter approximation can be employed in scenarios with \( K_{\text{min}}/K_{\text{max}} \to 0 \) to calculate an equivalent steepest decrease in using physical clogging \( (m_{\text{eq}}) \) as proxy for describing a biological clogging development. The only requirement for the quantitative comparison is knowing both \( K_{\text{max}} \) and \( b \).

**Fitting of the Function with Experimental Data**

The analytical clogging function was fitted to the data by minimizing an adapted version of the root mean square error (RMSE). Hereby, residuals were calculated in respect to the average of the actual measured hydraulic conductivity for each time \( t_i \). This adaptation was performed because hydraulic conductivities span over two magnitudes and deviations at lower hydraulic conductivities can be neglected. This is shown in Equation 9, that is,

\[
\text{RMSE}_{\text{rel}} = \sqrt{\frac{1}{T} \times \sum_{i=1}^{T} \left( \frac{K_{\text{1i}} + K_{\text{2i}}}{2} - K(t_i) \right)^2},
\]  

(9)

where \( K_{\text{1i}} \) and \( K_{\text{2i}} \) are the determined hydraulic conductivities for two identical models at times \( t_i \), \( K(t_i) \) is the result of the parametrized Equations 4 and 5, respectively, at these times, and \( T \) is the total number of measurements over time.

**Results**

For the sake of readability, the following analysis is solely based on the evolution of hydraulic conductivity. The pressure head evolution is to be found in Appendix S1.

**Physical Clogging**

Figure 4 shows the physical clogging development for experimental series \( A_{\text{phy}} \) and \( B_{\text{phy}} \). The difference becomes obvious by comparing the parameter \( m \) (Equations 3 and 4) as the relationship is proportional to the clogging speed. Considering the 63 mm SDW (no gravel pack; A63_nGP) in Experiment \( A_{\text{phy}} \), \( m \) is three times lower than for the half-sized 32 mm SDW (no gravel pack; A32_nGP), but 160 times higher than for the two effectively double-sized conventional wells with gravel pack (screen diameter: 32 mm/63 mm A120_GP (1) and (2)). In experiment \( B_{\text{phy}} \) the steepness factor \( m \) for the SDW (screen diameter: 63 mm; B63_nGP) is 40 times higher than for a well with the same effective well diameter due to the gravel pack (screen diameter: 32 mm; B63_GP). For the two gravel aquifer setups (B63/32_GA), however, no clogging was observed at all.

**Biological Clogging**

Figure 5 shows the biological clogging development for Experiments \( A_{\text{bio}} \) and \( B_{\text{bio}} \). The oscillations were most likely caused by flow rate variations but can also represent disturbances during the clogging process itself. Nevertheless, both the temporal and the general clogging behavior are still observable. The assessment of the differences is more sophisticated here than for physical clogging as both the inflection point \( c \) and the steepness \( b \) have to be considered (Equation 5). When comparing the 63 mm SDW (A63_nGP) in experiment \( A_{\text{bio}} \) with its half-sized, that is, 32 mm equivalent (A32_nGP) the inflection point is slightly delayed but steepness rises. For the two effectively double-sized conventional wells with gravel pack (screen diameter: 32 mm/63 mm; A120_GP), however, the inflection point remains similar but the steepness is reduced significantly by a factor of five. The 63 mm SDW (B63_nGP) in Experiment \( B_{\text{bio}} \) shows a slightly earlier inflection point. However, the steepness is three times higher than for conventional wells equipped with a gravel pack (screen diameter: 32 mm; outer diameter: 63 mm; B63_GP). For the gravel aquifer, finally, both parameters are in the same magnitude as for the other models. All models converged to a \( K_{\text{min}} \) (Equation 4), that was approximately two orders of magnitude lower than the initial hydraulic conductivity \( (K_{\text{max}}) \) such that the approximation of Equation 8 becomes valid.

**Discussion**

Parameters defining the fitted functions for physical and biological clogging are listed in Table 1. Since \( K_{\text{max}} \) is not dependent on the clogging itself, it can rather be considered an estimate of the initial \( K \) which seems appropriate for the sandy models, comparing the assessed \( K \)’s
(1.5–4.8 × 10⁻⁴ m/s) with the one determined beforehand in the permeameter test (2.0 × 10⁻³ m/s). The determined \( K_{\text{max}} \) for the gravel, however, seems rather low as \( K \) was estimated beforehand to be around 1 × 10⁻² m/s, using the method by Beyer (1964). However, this method is actually not suitable for gravel and the discrepancy can also be related to the barely measurable hydraulic head differences (∼1 mm) for the gravel increasing the uncertainty of the \( K_{\text{max}} \) assessment for the gravel.

A comparison of \( t_{10\%} \) values, that is, the times when clogging leads to a \( K \) drop down to 10% of \( K_{\text{max}} \), shows significant differences, especially for the physical clogging. It is highlighted that none of the conventional wells equipped with a gravel pack ever reached this critical value during the experimental period. In contrast, time differences during biological clogging in experiments ABio and BBio are in a very close range (factor of three and factor of two). Interestingly, the gravel aquifer appears to react in a similar way to the sand aquifers indicating that biological clogging was not delayed by the bigger pores of the medium. Well screen diameter, however, seems to have a similar effect on this parameter in both cases (A63_nGP vs. A32_nGP: factor 1.25 for biological and 1.5 for physical clogging).

Taking a more detailed look on \( K_{\text{min}} \) for biological clogging, it is consistently 1.5 to 2 orders of magnitudes lower than \( K_{\text{max}} \). This applies to all investigated setups. Small differences are notable as \( K_{\text{min}} \) tends to be slightly higher for wells with larger diameters and gravel packs but the differences are mostly in the range of measurement inaccuracy. Sometimes (especially for A120_GP) it is not even clear whether \( K_{\text{min}} \) has been reached at all. Also the inflection point \( c \) shows differences of 14.7 (Experiment ABio) and 9.1 h (Experiment BBio) which seems rather small compared to the duration of the biological clogging experiments.

As explained in Materials and Methods section, the steepness factors \( m \) and \( b \) of the physical and biological clogging can be compared directly to each other at their steepest points via a conversion of \( b \) into \( m_{\text{eq}} \) (see Equation 8). The differences in \( m_{\text{eq}} \) values for the biological clogging experiments are much smaller (e.g., A_120_GP vs. A32_nGP: roughly a factor of 12) than the respective differences between the \( m \) values for physical clogging (A_120_GP vs. A32_nGP: roughly a factor of 460). This indicates that the different well setups appear to have a smaller impact on the biological clogging experiments than on the physical clogging experiments. Notably so, the diameter of a well with no gravel pack seems to hardly affect \( m_{\text{eq}} \) (A63_GP vs. A32_GP: factor of 1.2). In contrast, the addition of a gravel pack appears to have a notable effect (B63_GP vs. B63_nGP: roughly a
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Table 1

|             | A120_GP | A63_nGP | A32_nGP | B63_GP | B63_nGP | B63-32_GA |
|-------------|---------|---------|---------|--------|---------|-----------|
| Physical clogging |         |         |         |        |         |           |
| $K_{\text{max}}$ (m/s) | $3.4 \times 10^{-4}$ | $3.6 \times 10^{-4}$ | $1.8 \times 10^{-4}$ | $1.5 \times 10^{-4}$ | $4.0 \times 10^{-4}$ | $5.6 \times 10^{-3}$ |
| Steepness $m$ (l/m) | 52.2    | 8200.0  | 24,300.0 | 671.0  | 27,500.0 | 0.0       |
| $t_{10\%}$ (h)     | –       | 3.08    | 2.08    | –      | 0.83    | –         |
| Biological clogging |         |         |         |        |         |           |
| $K_{\text{max}}$ (m/s) | $4.8 \times 10^{-4}$ | $1.8 \times 10^{-4}$ | $1.5 \times 10^{-4}$ | $3.0 \times 10^{-4}$ | $2.2 \times 10^{-4}$ | $2.0 \times 10^{-3}$ |
| $K_{\text{min}}$ (m/s) | $9.3 \times 10^{-6}$ | $6.0 \times 10^{-6}$ | $2.3 \times 10^{-6}$ | $6.3 \times 10^{-6}$ | $4.0 \times 10^{-6}$ | $2.1 \times 10^{-5}$ |
| Inflection point $c$ (h) | 24.3    | 38.1    | 23.4    | 26.2   | 17.1    | 25.0      |
| Steepness $b$ (−)  | 0.04    | 0.22    | 0.15    | 0.10   | 0.33    | 0.14      |
| Eq. steepness $m_{\text{eq}}$ (l/m) | 20.80   | 305.5   | 250.0   | 83.30  | 375.0   | 17.50     |
| $t_{10\%}$ (h)     | 130.75  | 50.25   | 40.25   | 51.75  | 24.5    | 42.25     |

The absolute reproducibility of the results seems limited, especially when A63_nGP and B63_nGP are compared. For physical clogging, this could be due to different stirring techniques in Experiment APhy and BPhy which might have caused differences in effective turbidity of the infiltrated water. For biological clogging, the range of parameters which were not monitored or mostly beyond control of the authors (temperature, oxygen levels, biocenosis in the reservoir, etc.) might have caused these differences.

During biological clogging experiments, flow tubes had to be repeatedly cleaned as microbial growth caused...
a decrease in flow rate. Nevertheless, flow rates dropped for few hours during the night, sometimes down to 2 mL/s. Furthermore, the circulation of the water in a closed loop with a constant addition of nutrients bears certain restrictions. The results would certainly be more realistic if a continuous supply with nutrient-rich water is arranged.

From a conceptual point of view, one could argue that comparing the two very different clogging processes is doubtful, even with simplified indicator values such as $t_{10\%}$ or $m/m_{eq}$. Hence, the comparison should not be overinterpreted. It should rather be seen as a qualitative and easy-to-measure indicator for the different impacts of well setups on the clogging processes. For any quantification of the differences, case-specific experiments need to be conducted under fully controlled and representative conditions.

Conclusions

The experimental setup used in this study proved to be suitable to simulate clogging in different well setups. Even though reproducibility was not completely demonstrated, the setup could be improved by better controlling the boundary conditions, though this seems especially demanding for biological clogging.

In future investigations, aquifer-well models like the ones used in this study could offer an easy-to-realize and, hence, low-cost solution to simulate clogging in MAR systems for longtime runs, to generate sets of estimate functions for clogging and to assess how different scenarios (intermittent infiltration, disinfection, regeneration) could affect the infiltration capacity for certain well setups. So far, the models have only been used for assessing short-term effects under rather synthetic conditions. Nevertheless, they could be easily set up and applied (even on-site) to assess different real-world well setups by using a specific kind of water that is to be recharged.

The results of the experiments confirm the intuitive relationship described in Figure 1: Both, the comparisons of $t_{10\%}$ values and the comparison of $m$ with $m_{eq}$ show a slight delay in clogging for SDWs with a larger diameter and a bigger delay for conventional wells with a gravel pack. However, both effects seem significantly smaller for biological clogging than for physical clogging. Therefore, the actual benefit of a more sophisticated (and expensive) conventional, large-diameter well setup should be evaluated prior to construction. Otherwise, a well-based MAR project might be dismissed too quickly due to disproportionately high costs, even though a solution with SDWs could offer a similar benefit with substantially lower expenses.

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Author Contributions

F.K., M.B., and F.H. conceived and designed the experiments; F.K. and L.G. performed the experiments; F.K. analyzed the data; F.K., M.B., F.H., and R.L. wrote the paper; F.K., M.B., F.H., and R.L. revised the paper.

Authors’ Note

The authors do not have any conflicts of interest or financial disclosures to report.

Supporting Information

Additional supporting information may be found online in the Supporting Information section at the end of the article. Supporting Information is generally not peer reviewed.

Appendix S1. Supporting Information.

Figure S1. Pressure Head (a) & Resulting K (b) for Experiment A (Physical Clogging)
Figure S2. Pressure Head (a) & Resulting K (b) for Experiment B (Physical Clogging)
Figure S3. Pressure Head (a) & Resulting K (b) for Experiment A (Biological Clogging)
Figure S4. Pressure Head (a) & Resulting K (b) for Experiment B (Biological Clogging)

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