Groundwater Monitoring&Remediation

Column

Advances in Remediation: Recharge *Redux*

by Craig Divine, Charles Newell, Kenneth Walker, Matthew Spurlin, Timothy Blount, Prashanth Khambhammettu and Marc Killingstad

Introduction

In our first undergraduate hydrogeology course, we learned that, according to the inviolable Darcy's Law, groundwater flow is driven by potential energy, expressed as the hydraulic gradient. However, viewed another (and we propose possibly a more useful) way, the hydraulic gradient observed/measured at sites is instead a *response* to the resistance to flow through aquifers that have finite permeability. For systems that exhibit "homeostasis" (i.e., dynamic equilibrium), recharge—the rate of water entering the system, or the "mass flow rate in"—is therefore a fundamental input to groundwater flow, as shown in the conceptual mass balance in Figure 1. In this article, we consider recharge not as the formal potential-energy driver of groundwater flow, but as a boundary-condition input that strongly influences the magnitude, timing, and spatial distribution of groundwater flux.

While hydraulic gradients are straightforward to measure, viewing recharge as a fundamental control on groundwater flow helps explain why groundwater velocities in many porous-media water-table aquifers span a much narrower range than geologic permeability alone would suggest. Hydraulic conductivity can vary by eight or nine orders of magnitude across unconsolidated deposits, but groundwater seepage velocities at contaminated sites commonly fall within a much smaller practical range; in our experience, frequently between about 50 and 1000 ft/year. Consistent with this, the Hydrogeologic Database (HGDB), developed from a nationwide survey of groundwater professionals conducted through the National Water Well Association, includes estimated groundwater seepage velocities from 290 remediation sites across the United States. Approximately 70% of those velocities ranged from about 6 to 15,000 ft/year (Newell et al. 1990). This range broadly tracks regional differences in precipitation and recharge across the continental United States, reinforcing the point that recharge, rather than permeability alone, is often the controlling system input for groundwater flux in water-table aquifers. Figure 2 shows a conceptual water mass balance for several aquifers with varying permeability and recharge characteristics, further highlighting the relationship between recharge, groundwater flux, and observed hydraulic gradient.

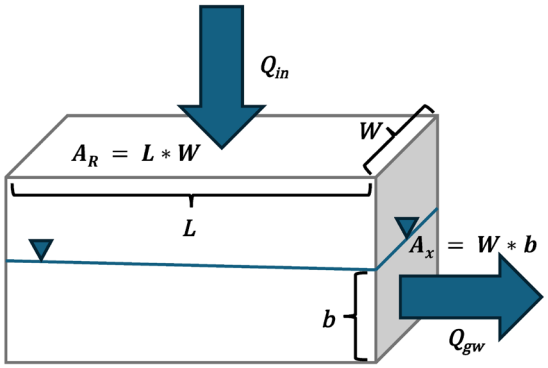
Over the past couple of decades, substantial investment in high-resolution site characterization technologies has improved our ability to define local subsurface heterogeneity, which appropriately serves as the roadmap for site-scale contaminant hydrogeology. However, that same level of attention has not been given to better defining the spatial and temporal variability associated with groundwater recharge. Consider too that the subsurface structure and properties are generally only spatially variable, whereas recharge varies over space *and* time.

The key point we are making here is this: understanding the spatial and temporal variability of recharge (natural or anthropogenic) is fundamental to our aquifers of interest because recharge is a principal water-budget input that sustains groundwater flow and, consequently, strongly influences contaminant transport. This is intuitive and has been long understood—Darcy (1856), Slichter (1899), Theis (1940), Tóth (1962), and many other early hydrogeologists all clearly recognized and discussed the central role of recharge in groundwater-flow dynamics. Yet how often do we measure, quantify, explicitly acknowledge, or even attempt to characterize recharge in our conceptual site models (CSMs)?

Many classical approaches to recharge characterization, like hydrogeology more broadly, were initially motivated by groundwater-resource questions at larger spatial and temporal scales: How much water enters an aquifer in a year or decade? How much groundwater can be sustainably produced? How rapidly are groundwater resources replenished? While these questions remain important for water-resource management, contaminant hydrogeology requires a finer-scale understanding of where recharge occurs, when it occurs, and what contaminant mass recharge water encounters along its path to the water table. Persistent contaminant sources in the unsaturated

Article impact statement: Recharge is a fundamental control on groundwater flow and contaminant mass flux, and its spatial and temporal variability deserves greater emphasis in contaminated-site conceptual models. A practical framework is provided for incorporating recharge into source prioritization, mass-discharge estimates, and remedial decision-making, with particular relevance to PFAS sites.

© 2026 National Ground Water Association.
doi: 10.1111/gwmr.70090
All rights reserved, including rights for text and data mining and training of artificial intelligence technologies or similar technologies.



$$(1) \quad Q_{gw} = Q_{in}$$

$$(2) \quad Q_{in} = R * A_R = R * W * L$$

$$(3) \quad q = Q_{gw}/A_x = Q_{gw}/(W * b)$$

Substituting (1) and (2) into (3) gives:

$$(4) \quad q = R * L/b$$

Figure 1. Conceptual mass balance in a recharge-dominated system in equilibrium. In this figure the left boundary represents the groundwater divide (i.e., no groundwater flow is entering from the left). Using continuity (Equation 1), and dimensional analysis (Equations 2 and 3), groundwater flux (also referred to as "Darcy velocity") can be expressed as a function of recharge (Equation 4), independent of the hydraulic gradient. In unconfined systems, aquifer thickness will vary according to the rate of recharge (i.e., flow in) and soil hydraulic conductivity (i.e., resistance to flow). (Q = flow out of system or groundwater flow [length cubed per time]; Q_{in} = flow into system due to recharge [length cubed per time]; R = average recharge [length per time]; A_R = catchment area [length squared]; W = catchment width [length]; L = catchment length [length]; A_x = aquifer cross-sectional area [length squared]; b = aquifer thickness [length]; q = groundwater flux [length per time]).

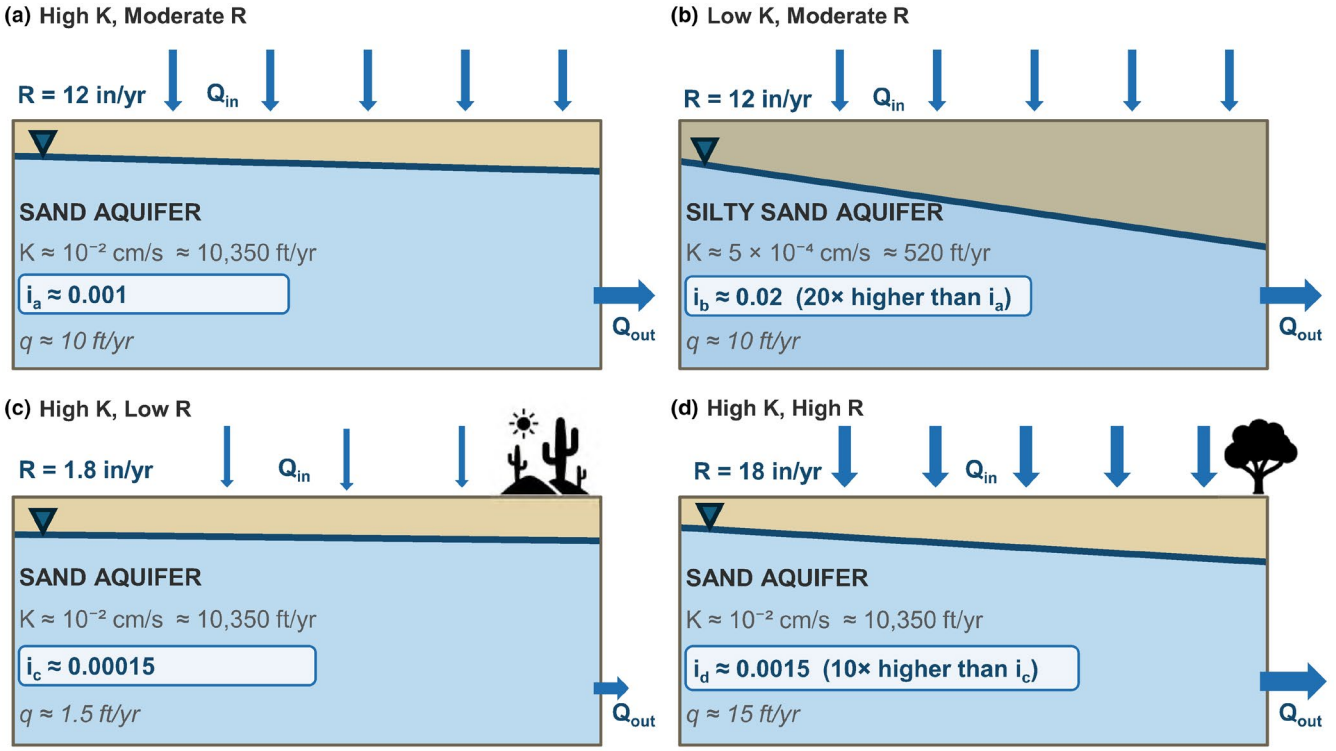


Figure 2. Conceptual mass balance on four aquifers with varying permeability and recharge characteristics. Temporal variation in aquifer thickness is assumed to be small relative to variation in recharge, and hydraulic gradients are exaggerated for visual clarity. With recharge fixed, hydraulic gradient increases by a factor of 20 relative to panel (a) in response to a 20-fold decrease in hydraulic conductivity in panel (b), while groundwater flux remains unchanged. Panels (c) and (d) illustrate variation in recharge for arid and humid climates, respectively, with a 10-fold increase in recharge producing a corresponding 10-fold increase in hydraulic gradient and groundwater flux. Input parameters and calculations are presented in Table S1.

zone can be episodically mobilized by recharge over both time and space. As a result, recharge should be considered not merely as a regional water-budget term but as a dynamic control on contaminant loading, mass flux, and mass discharge.

Mass flux and mass discharge concepts have been applied to conventional groundwater contaminants for two decades, but they take on heightened importance for PFAS. Unlike petroleum hydrocarbons and chlorinated solvents, many PFAS—especially the perfluoroalkyl acids (PFAAs) of regulatory concern—do not naturally break down in the environment to any significant degree. At AFFF release sites, this persistence makes recharge a critical data gap for quantifying PFAS mass flux and mass discharge through the vadose zone to groundwater (e.g., Newell et al. 2023).

Recharge must also be considered at finer spatial and temporal scales because PFAS retention and mobilization in the vadose zone are strongly influenced by transient soil-moisture conditions and multi-phase processes, including solid-phase sorption and air-water interfacial (AWI) adsorption/desorption (Costanza et al. 2019; Schaefer et al. 2019; Sharifan et al. 2021; Brusseau and Guo 2023; Divine et al. 2024; Russell et al. 2025; Kolade et al. 2026). Note that in this article, the terms unsaturated zone and vadose zone are generally used interchangeably; however, the unsaturated zone is defined strictly as the zone where water saturation is less than 100%, whereas the vadose zone describes the region above the water table where water pressure is less than atmospheric pressure, including portions of the capillary fringe where water saturation may be 100%.

As depicted in the two left conceptual images in Figure 3, the collapse of the AWI area during infiltration events can mobilize sequestered PFAS. As shown in the far-right panel in Figure 3, this phenomenon produces dynamic, time-variable changes in porewater concentrations rather than representing a steady-state condition (Guo et al. 2020; Newell et al. 2023; Stults et al. 2024). Recent field monitoring reported by Kolade et al. (2026) provides direct support for this recharge-controlled conceptual model. At an AFFF-impacted firefighting training site, two years of continuous vadose-zone monitoring showed that PFAS porewater concentrations increased during wet periods with elevated infiltration and unsaturated-zone water content, whereas long-chain PFAS, particularly PFOA and PFOS, decreased markedly during drier periods, consistent with stronger retention under lower-moisture conditions.

These processes reinforce the importance of understanding not only where but when recharge occurs within the vadose zone and how the effects vary with depth. Above the zero-flux plane (ZFP), the elevation plane where the vertical hydraulic gradient is zero, infiltration and evapotranspiration produce pronounced fluctuations in soil moisture, with periods of rainfall promoting downward transport and dilution and periods of evapotranspiration promoting upward water flux and evapoconcentration (Healy 2010). As a result, porewater concentrations can be highly dynamic or "flashy" (Guo et al. 2020; Wallis et al. 2022; Newell et al. 2023). Below the ZFP, soil moisture is generally less affected by vegetative uptake and surface evaporation, resulting in comparatively stable downward drainage and recharge conditions (Allison et al. 1994; Shah et al. 2007; Healy 2010; Newell et al. 2023).

There is also evidence suggesting that soil-moisture fluctuations in arid, semi-arid, and some temperate climates may govern PFAS mobility more strongly than static soil properties alone (Schaefer et al. 2021, 2022a; Zeng and Guo 2021; Vahedian et al. 2024). The timing, magnitude, rate, and spatial distribution of recharge (e.g., focused vs. diffuse recharge) provide the context for improved understanding of PFAS mobilization, how partitioning to porewater responds, and ultimately how contaminant mass migrates to groundwater. Conceptually, PFAS behavior in the vadose zone has the potential to be highly dynamic, and field observations at some sites support this view. However, the data needed to characterize these dynamics remain limited and are not routinely collected. We suggest that recharge may need to be characterized and monitored at much finer temporal and spatial resolution than is typical under current standard practices, rather than treated as an averaged hydrologic parameter.

With thousands of AFFF release sites, persistent PFAS constituents, and remediation technologies that remain costly and immature, practitioners need a defensible, flux-based framework to prioritize which sources warrant remediation priority and to evaluate the actual risks that

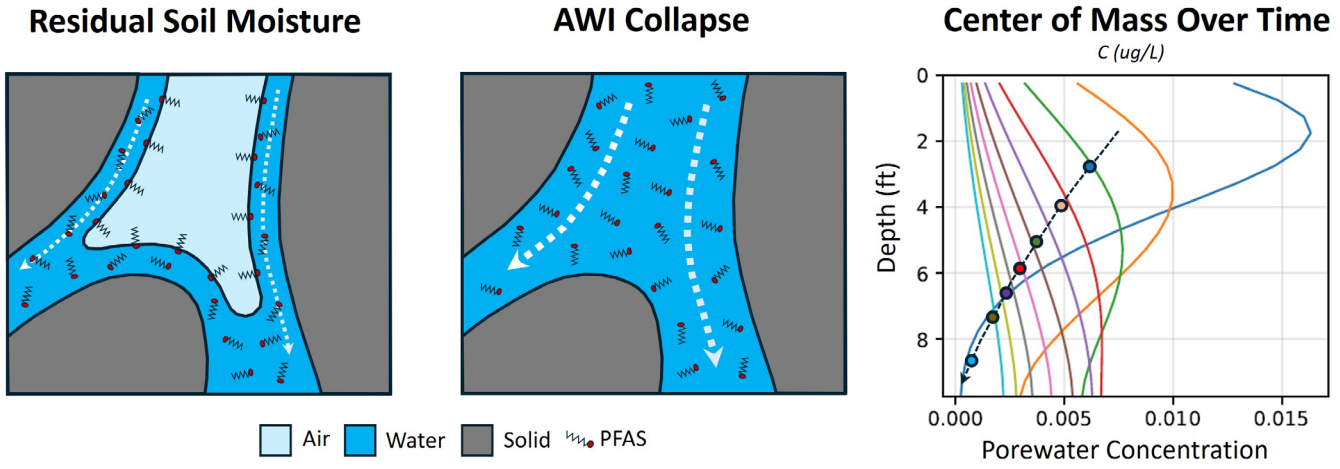


Figure 3. Soil moisture fluctuations are a dominant control on the mobility of many PFAS compounds. As shown in the left panel, between recharge events, some PFAS accumulate at the air-water interface because of their hydrophilic head and hydrophobic tail structures. During strong recharge events, the air-water interfacial area can collapse, mobilizing PFAS compounds into porewater, as shown in the center panel. These "recharge waves" can therefore produce significant pulses of mass loading to groundwater compared with intervening periods, when the air-water interface effectively sequesters PFAS in the unsaturated zone. Over time, repeated recharge events drive the PFAS center of mass downward toward the water table while reducing overall porewater concentrations in the unsaturated zone. PFAS flushing will be slowed as it encounters residual trapped air. This progression is illustrated in the right panel, where the lines show porewater PFAS concentration profiles over time, represented by different colors. The circles indicate the average PFAS concentration and the location of the center of mass at each time, with circle colors matching the corresponding concentration profiles.

individual sources pose to downgradient receptors. We believe that recharge—specifically, its spatial and temporal distribution—is not only undervalued in the world of contaminant hydrogeology, but it is the critical input that allows a mass flux-based framework to be more quantifiable, precise, and effective for CSM development and refinement and, ultimately, site management.

In this column, we are calling for a "recharge *redux*"—a renewed and modern focus on recharge—to remind the remediation community that recharge is a primary input and organizing control in contaminant hydrogeology. To complement the written text, we have recorded an audio summary that is included as downloadable Supplemental Information where we summarize the main features of this article and discuss what we consider key takeaways and implications (Audio S1).

What Is Recharge and Why Does It Matter at Contaminated Sites?

Recharge is simply the water that infiltrates into the soil and percolates below the zone of evapotranspiration, reaching the underlying aquifer system or returning to the surface as base flow or discharge in streams and springs. Few available methods provide direct measurements of recharge; most recharge estimates are developed from indirect approaches such as water budgets, inverse modeling, surface-water methods, lysimetry, in situ soil-moisture and matric-potential sensors, environmental tracers, water table fluctuations, and geophysics (e.g., Brutsaert 1982; Alley 1984; Lerner et al. 1990; Ronan et al. 1998; Sanford et al. 2002; Healy and Cook 2002; Delin and Herkelrath 2007; Goebel and Knight 2021).

Because of the uncertainties associated with each of these approaches, the range and reliability of recharge estimates can vary widely, reflecting the complexity of this vital hydrologic component and its impacts on contaminant hydrogeology (Scanlon et al. 2002). Recharge magnitude varies in space and time, leading to variable distributions of contaminant concentrations in the subsurface. It is, as John Bredehoeft has pointedly indicated, the most difficult component of the groundwater system to quantify (e.g., Bredehoeft et al. 1982; Bredehoeft 1997, 2007). Because many contaminated sites are physically small relative to typical catchment scales, and because contaminant sources and plumes often have localized characteristics, environmental remediation practitioners have historically characterized recharge and vadose-zone contaminant mobilization using broad, simplified assumptions.

Challenge of Temporal Variation in Recharge

Consider three hypothetical sites—A, B, and C—that have similar total annual recharge but differ in how and when recharge occurs. Site A might be in the eastern United States, where recharge occurs through precipitation events distributed throughout the year; vadose-zone sources may be mobilized relatively continuously. Site B, located in the Midwest, may have a seasonal pattern characterized by frozen ground during winter, spring snowmelt, and summer storms; contaminant mobilization in the vadose zone would likely be episodic. Site C, in the Intermountain West, may experience long periods with little direct precipitation recharge, with most recharge derived from spring snowmelt and mountain-front recharge. Even where annual recharge totals are similar, spatial and temporal variability can produce very different groundwater responses and contaminant transport behavior. These differences directly affect how hydrogeologists and remediation practitioners develop CSMs, design sampling programs, and evaluate remedial approaches.

While annual averages may be sufficient for groundwater-resource studies, they can conceal important processes affecting contaminant migration through the subsurface, particularly through the vadose zone. Where the air-water interface controls PFAS sorption and mobility (Costanza et al. 2019; Schaefer et al. 2019, 2021, 2022a; Guo et al. 2020; Sharifan et al. 2021; Zeng and Guo 2021; Wallis et al. 2022; Brusseau and Guo 2023; Vahedian et al. 2024; Russell et al. 2025; Kolade et al. 2026), event-scale recharge may be especially important. For example, some PFAS vadose-zone sources exhibit a "first flush" response, with elevated porewater concentrations during recharge events.

Annual or longer timescales (e.g., inches per year) are generally adequate for water-supply and resource management; drought planning, for example, often relies on multi-year or decadal analyses. Short-term variability in recharge, which may matter little for these water-resource assessments, is critical for contaminant transport. Figure 4 illustrates this concept with soil-moisture data collected over time at a site in North Dakota. The precipitation-driven recharge response is most pronounced at the shallowest sensor, becomes attenuated with depth, and shows that not every precipitation event produces recharge. Interestingly, as shown in Figure 5, soil moisture was also measured at this site using downhole nuclear magnetic resonance at a single location during four monitoring events over a 1-year period. Those results indicate that, during monitoring events not associated with precipitation, moisture conditions remained relatively steady through time. This behavior has led some researchers working in arid environments to propose a threshold precipitation amount below which recharge does not occur. Even that threshold is partly controlled by antecedent soil-moisture conditions, which is why soil matric-potential sensors are often more informative than soil-moisture sensors alone in drier settings.

At many sites, dynamic recharge produces short-term changes in groundwater levels, hydraulic gradients, and groundwater velocity. This response can be nearly instantaneous in permeable settings with shallow water tables and in many fractured rock and karst systems (e.g., Worthington 2026). At other sites, recharge-driven water-level changes are more muted and expressed primarily as seasonal trends. Figure 6, for example, shows water-level and calculated hydraulic-gradient data from a northern Michigan site where spring snowmelt dominates recharge, producing a seasonal trend in hydraulic gradient and inferred groundwater velocity.

These temporal changes have important contaminant-transport implications. For chemicals with non-equilibrium sorption–desorption behavior or other kinetic controls, including PFAS (e.g., Schaefer et al. 2022a, 2023), the rate and timing of recharge fronts can affect how much contaminant mass

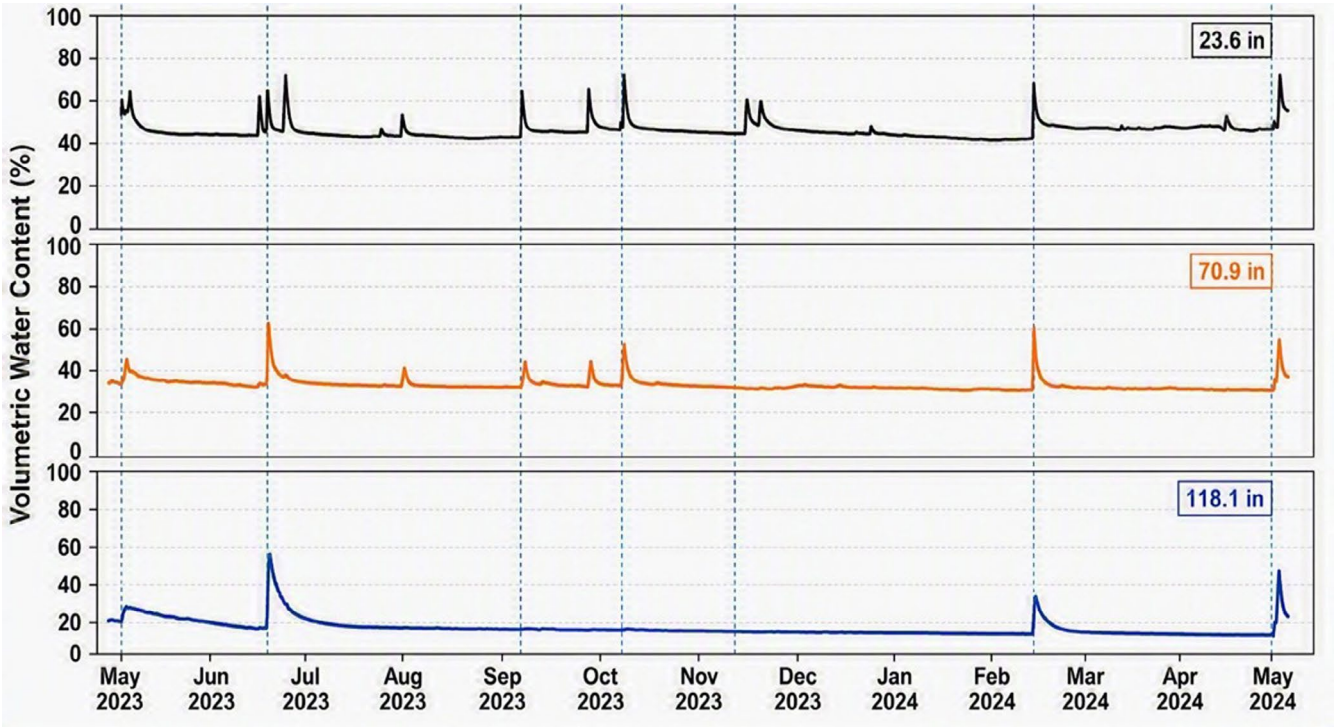


Figure 4. Soil moisture responses from probes installed at three depth intervals: 23.6 inch below ground surface within the treated soil (black line), 70.9 inch below ground surface below the treated soil (red line), and 118.1 inch below ground surface below the treated soil (blue line). Spikes indicate increased soil moisture associated with discrete precipitation events. For some events, responses are detected at all three depths at approximately the same time, indicating rapid infiltration.

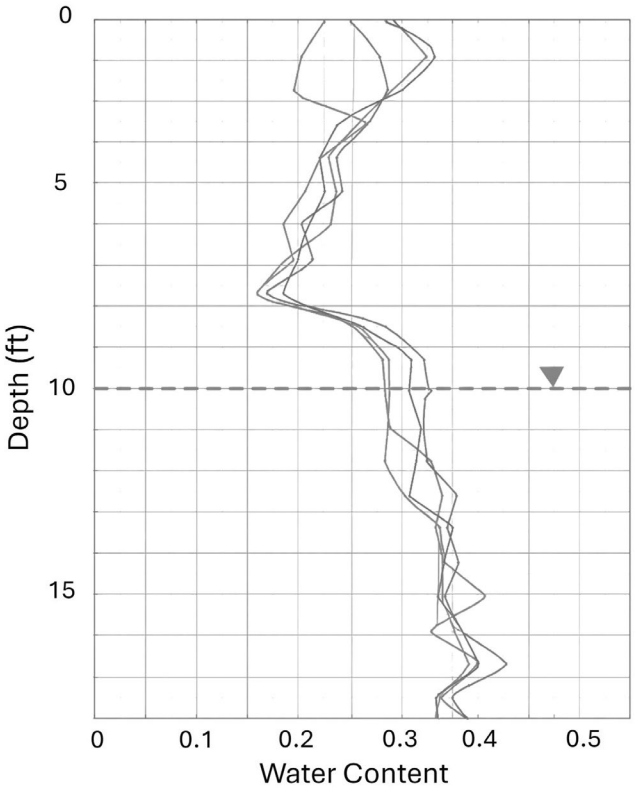


Figure 5. NMR imaging data from the same site as Figure 4 at a single location during four monitoring events over a 1-year period. Results indicate that during monitoring events not associated with precipitation, moisture conditions remained relatively steady through time.

is mobilized through the vadose zone and into groundwater. Rapid vadose-zone flow may limit PFAS desorption into recharging porewater, producing lower groundwater concentrations than slower, more persistent recharge for the same total recharge volume. Direct groundwater-velocity measurements from point velocity probes further show that groundwater velocity can be both spatially and temporally variable at many sites (e.g., Labaky et al. 2009; Kempf et al. 2013; Osorno et al. 2018, 2022; Heyer et al. 2021, 2022).

For sites where contaminant mass flux is an important component of the CSM, as it is at most PFAS sites, recharge should be evaluated on two timescales: (1) the event scale, spanning several days, which is the preferred window for collecting porewater samples; and (2) the annual scale, which supports annual mass discharge calculations. To identify when a recharge event large enough to warrant sampling is underway, a recent study recommended that precipitation records alone will likely suffice at humid sites, whereas soil matric-potential sensors proved useful at arid sites for timing porewater sampling (Walker et al. 2026). Once representative porewater concentrations are available, they can be multiplied by recharge values to determine mass discharge from the vadose zone to groundwater. Relying on precipitation data alone, at least in arid environments that exhibit more flashy behavior, may not be sufficient to capture the recharge front or identify optimal porewater sampling windows.

Challenge of Area-Averaged Recharge Estimates

Because tools for reliably estimating and characterizing the spatial distribution of recharge remain limited, practitioners have historically assumed simplified distributions using

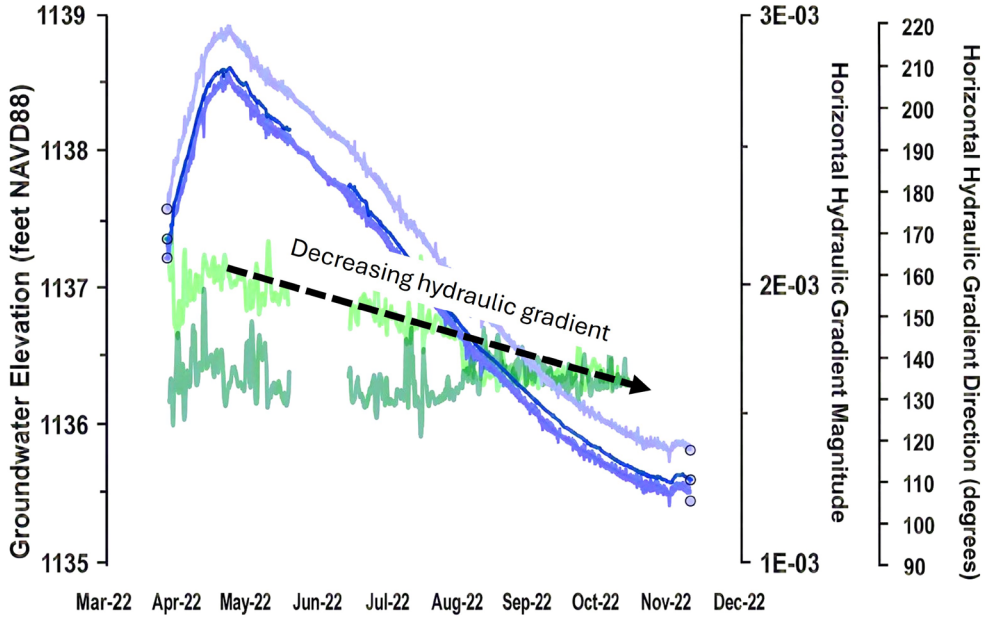


Figure 6. Seasonal recharge-driven variability in groundwater levels (blue series), groundwater flow direction (dark green series), and horizontal hydraulic gradient magnitude (light green series) at a site in northern Michigan. The hydrographs (blue series) indicate a pronounced spring snowmelt-dominated recharge response that, after the hydrograph peak, results in a seasonal decrease in hydraulic gradient and therefore a corresponding decrease in inferred groundwater velocity over the year. Groundwater velocity is not constant. Figure created by Trey Fortner.

area-averaged recharge rates, often at the regional scale. Recharge rates are spatially heterogeneous, and although this variability may be less critical for water-resource assessments, it is an important consideration for contaminant hydrogeology. Areas of focused recharge and preferential flow can drive rapid contaminant migration through the vadose zone to underlying aquifers. This is especially true in natural environments with karst geology, caliche layers, or arid climates where high-precipitation seasonal monsoon events produce surface-water runoff to infiltration basins. Focused recharge has also been identified in urban settings where land cover is often assumed to be relatively impervious, as illustrated by the water-leak example shown in Figure 7. Runoff from large paved areas may be directed to centralized infiltration areas; aging concrete and asphalt can create discrete zones of enhanced recharge, and leaking water, sanitary, or stormwater infrastructure can be a significant source of recharge in highly developed or industrialized areas. Recharge studies for contaminant transport should therefore evaluate whether recharge is primarily diffuse, allowing use of conventional estimation methods, or dominated by focused recharge processes that require different methods.

To protect groundwater resources, remedial efforts should be guided by contaminant mass flux and mass discharge, not solely by the distribution of contaminants in soil. Applying a flux-based framework to design and operate an effective remedial system therefore requires a local-scale understanding of recharge distribution. At some sites, it may be more effective to leave higher-concentration soil in place while focusing excavation or surface capping on lower-concentration areas that receive greater recharge, because those areas may contribute more strongly to groundwater contaminant loading. To improve CSMs and optimize remedies, future research should apply readily available tools to better characterize local-scale spatial variability in recharge.

Established Recharge Measurement and Estimation Techniques

A simple Google Scholar search shows that since Meinzer and Stearns (1929) first attempted to estimate monthly recharge rates, numerous and diverse methods have been developed and applied to quantify recharge at various temporal and spatial scales, ranging from hourly to decadal and millennium-scale estimates, and from less than a square meter to catchment-scale estimates. One of the oldest and simplest techniques is the water table fluctuation (WTF) method (Meinzer and Stearns 1929), which is based on the premise that water-level increases in unconfined aquifers are due to recharge arriving at the water table. This method is described in detail in Healy and Cook (2002), and an example application is presented in Figure 8. As with all currently available techniques, however, the WTF method has important limitations. First, it provides only a spatially averaged recharge estimate. Second, as conceptualized, the method is best applied in regions with shallow water tables that are responsive to precipitation events, and it assumes that water-level fluctuations are caused by recharge rather than nearby pumping, atmospheric-pressure changes, entrapped air, or other phenomena. When applied under appropriate conditions and with consideration of these limitations, the WTF method can provide valuable insight and guidance for further investigations.

Scanlon et al. (2002) and Healy (2010) provide excellent summaries of the commonly applied methods, their implementation, the pros and cons of each method, and guidance

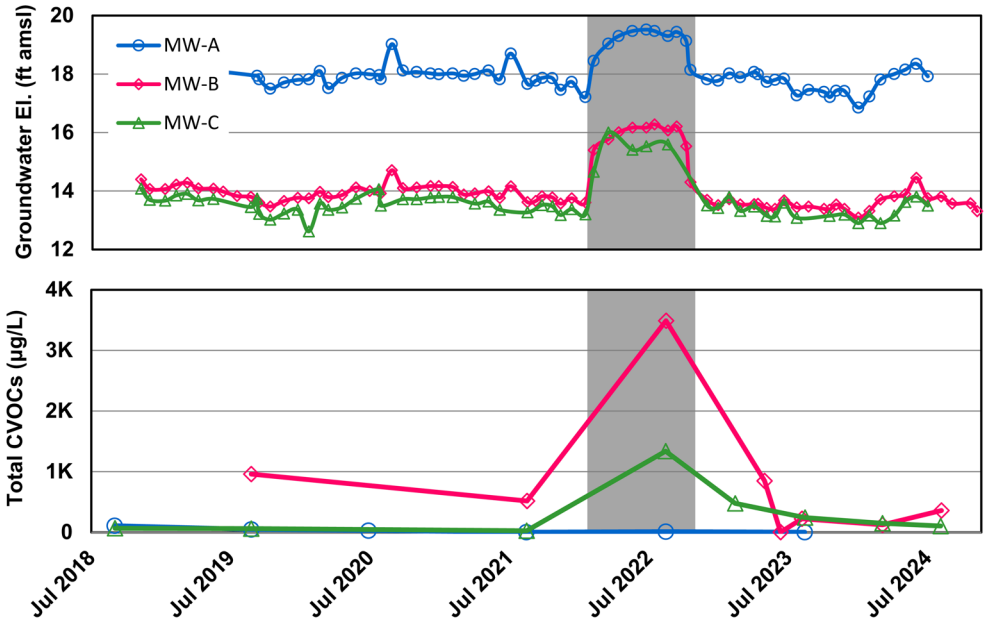


Figure 7. Top: Groundwater hydrograph data from monitoring wells upgradient (MW-A), 50 feet downgradient (MW-B), and 400 feet downgradient (MW-C) from a chlorinated volatile organic compound (CVOC) source area at a completely paved urban site. Groundwater mounding of up to 3 feet was observed during a persistent water leak, shown by the gray rectangle. Bottom: CVOC concentrations increased by factors of 3 and 20 at downgradient monitoring wells MW-B and MW-C (both screened from 25 to 35 feet below ground surface), respectively, following the leak and returned to baseline conditions over a period of 2 years.

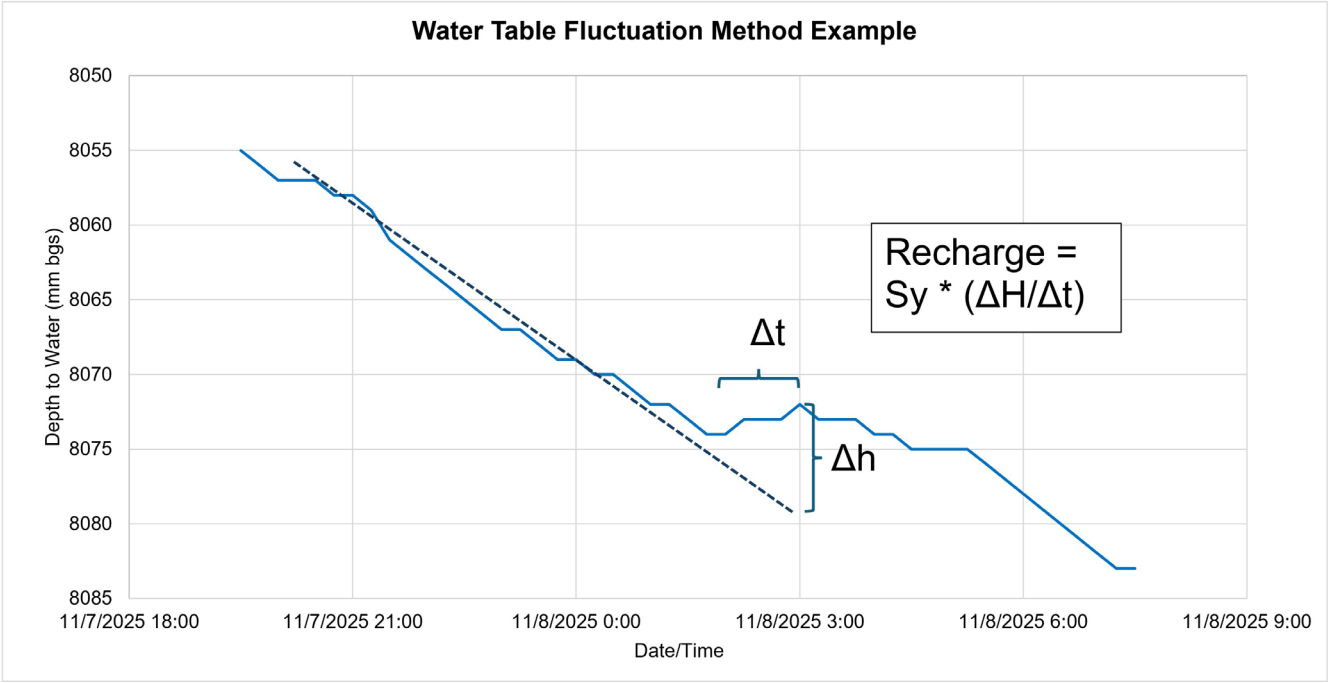


Figure 8. Example water level data (solid blue line) demonstrating use of the WTF method. The blue dashed line represents the trend of water table decline. In this example, the change in water table height from the extrapolated trend line to the top of the water level rise (Δ*h*) is approximately 6.3 mm over approximately 75 min (Δ*t*). When multiplied by the estimated specific yield (Sy) of the clay medium (0.02), this translates to 0.13 mm of total recharge over the 75-min recharge event.

for selecting the appropriate method(s) for a given set of site conditions. Most importantly, though, these researchers stress that (1) recharge estimation should be an iterative process with continual refinement as additional information becomes available and (2) multiple techniques should be applied (as appropriate) because of the inherent uncertainties in each method (i.e., multiple lines of evidence).

More recently, Newell et al. (2023) summarized these methods and identified a subset most applicable for estimating mass flux and mass discharge at a typical AFFF release

site. They grouped the methods into Tier 1, Tier 2, and Tier 3 categories based on site complexity and method cost and complexity. Newell et al. (2023) provide additional details on these methods and how to implement them.

As part of a recently completed Environmental Security Technology Certification Program (ESTCP) project (ER22-7381), several of these methods were reviewed in the context of estimating mass flux and mass discharge at two demonstration sites impacted by historical AFFF releases: one semi-arid site in Wyoming and one humid site in Tennessee (Walker et al. 2026). The results of 11 recharge estimation techniques tested at the two demonstration sites are presented in Table 1. At an annual scale, the Tier 1 methods produced internally consistent results at both sites, whereas several Tier 2 and Tier 3 methods yielded unreliable or unrealistic estimates that were excluded from further analyses. Tier 1 recharge estimates alone yielded the same estimated mass discharge as the average of the reliable Tier 1, 2, and 3 estimates at both sites. A key finding from this study was that simple, inexpensive empirical recharge methods may be sufficient for PFAS mass discharge estimation at many PFAS vadose-zone sites, including AFFF-impacted areas.

As described above, remote monitoring of recharge events with telemetered tensiometers, soil moisture probes, and soil matric-potential sensors can also provide key insights into the timing of recharge, especially in dry, arid to semi-arid soils. In these drier environments, not all precipitation events lead to recharge events, especially during hot summer months when evapotranspiration rates are highest, and these data can be utilized to understand the timing and depth of the recharge front. Matric potential measurements provide an indication of vadose zone moisture conditions and water mobility. Moni-

Table 1

Comparison of Recharge Estimates Derived from Empirical Methods, a Water-Mass-Balance Method, Drain-Gauge Lysimetry, Water-Table Fluctuations, and Tracer-Based Methods at Two AFFF-Impacted Demonstration Sites (Walker et al. 2026)

Method No.		Semi-Arid Site in/Year	Humid Site in/Year	Interpretation/Comments
1-A	Empirical regression: precipitation vs. recharge, based on data from Stephens et al. (1985)	0.9	14	Reliable at both sites
1-B	Nationwide high-resolution recharge map based on baseflow (Reitz et al. 2017)	0.7	11	Reliable at both sites
1-C	DRASTIC (Aller et al. 1985)	0–2 (avg. 1.0)	7–10 (avg. 8.5)	Reliable at both sites
2-B	Water-table fluctuation (WTF; see Figure 8 for example)	5.1	11	Humid sites only
2-C	Temperature tracer	–8,600	–130	Considered unreliable at either site; unable to isolate a temperature signal related to water transport from other processes, such as evaporation in the shallow subsurface
2-D	Unsaturated Darcy's Law with unit gradient	2,500	1	Considered unreliable at either site; no evidence of unit-gradient conditions at either site
2-E	Unsaturated-zone meteoric chloride tracer	0.08	>1.3	Considered unreliable at either site; multiple factors limited method applicability
2-F	Saturated-zone meteoric chloride tracer	0.02	1	Considered unreliable at either site; multiple factors limited method applicability
3-A	Unsaturated-zone water balance	0.7	—	Recommended at semi-arid and arid sites
3-B	Drain gauge lysimeter (DGL)	3.0	1,700	Installation artifacts
3-E	Groundwater tracer age-dating	1.1	Undetermined	Strong match at the semi-arid site

	Semi-Arid Site in/Year	Humid Site in/Year
Tier 1 recharge estimate average	0.9	11
Average of Reliable Tier 1, 2, 3 Estimates	0.9	11.2
Range	0.7–1.1	8.5–14
Coefficient of variation	0.2	0.21
Precipitation (inches per year)	15	57
Recharge/precipitation (%)	6%	19–20%

Note: Recharge estimates shaded in red represent values that were not deemed realistic and were excluded from further recharge calculations. Method numbers (e.g., 3-A) correspond to the methods specified in Table 3 of Newell et al. (2023).

toring temporal changes in matric potential can help identify periods immediately following infiltration or recharge when porewater is most accessible for sampling and when contaminant transport is likely to be most active (and PFAS concentrations are likely to be highest), thereby improving the timing and interpretability of porewater sampling events. In addition, when collecting porewater samples from suction lysimeters, soil matric-potential data can provide vital information on the optimal timing for porewater sampling events—whether sufficient porewater is present to pull into the lysimeters under vacuum and whether adequate sample volumes can be recovered for laboratory analysis.

Developing Techniques and Needs

The heightened focus on recharge at PFAS/AFFF sites has revealed several areas where improved methods for quantifying recharge at the local scale are needed. These include techniques for estimating recharge through engineered surfaces such as concrete and asphalt, and approaches for identifying sites where focused recharge, rather than diffuse recharge, is a significant component of the PFAS conceptual site model (CSM). Each of these topics is discussed in more detail below.

Recharge through Concrete and Asphalt: A Dual-Porosity Problem

Recharge through concrete and asphalt is now recognized as an important pathway at PFAS sites, yet little research has been published regarding how these engineered surfaces influence the distribution and magnitude of recharge. Practitioners routinely treat concrete and asphalt as a no-flow (impervious) upper boundary, on the assumption that pavement eliminates infiltration and, therefore, vertical mass discharge does not occur under these impervious surfaces. The available evidence, though, does not support that assumption at all sites, in part because concrete is not one porous medium but two.

Like fractured bedrock, concrete is properly conceptualized as a dual-porosity material (Douglas et al. 2023). Macroporosity comprises cracks, construction and expansion joints, and material interfaces, which occupy a small fraction of the surface area but can dominate infiltration much like fractures dominate flow in competent rock (Wiles and Sharp 2008; Salt and Kjeldsen 2019). Microporosity is the capillary pore network within the cement paste, which governs moisture retention and slows contaminant release across most of the slab volume (Neville 2011). Macroporosity allows water and dissolved mass to rapidly infiltrate during storm events, whereas microporosity stores mass and releases it over years to centuries. The rate and overall volume of recharge moving through the macroporosity of concrete and asphalt surfaces can potentially be significant. For example, Wiles and Sharp (2008) estimated that approximately 21% of mean annual rainfall in Austin, Texas, was potentially available for recharge to the subsurface through ostensibly impervious cover material.

Evaluation of cores collected under Strategic Environmental Research and Development Program (SERDP) project ER23-3713 suggests that flow through the macroporosity of AFFF-impacted pads may be more important than commonly assumed. At three facilities with strongly contrasting climates and source strengths, elevated PFAS concentrations were measured in the deepest interval sampled (typically 4 to 8 inch below the concrete surface) in five of six cores. One plausible explanation is that PFAS-impacted runoff, generated during fire training or subsequent storm events, moved rapidly toward and downward through joints and cracks to the base of the slab and then spread laterally along the slab-soil interface, a process referred to here as "underflow." In this scenario, underflow would deliver PFAS mass to the underside of the pad and the underlying soil across an area well beyond the individual joints and cracks through which it entered, effectively converting the bottom of the slab into a distributed source. Concentrations were also elevated adjacent to vertical joints and fractures, suggesting that these features may act both as preferential pathways and as vertical plane sources. While this dataset is limited, these observations suggest that, without site-specific data indicating otherwise, an underflow component should be considered at PFAS-impacted pads as part of CSM development.

Estimating vertical mass discharge at PFAS-impacted sites with concrete or asphalt pavement may therefore require resolving at least three fluxes: macropore infiltration through joints and cracks; lateral underflow along the bottom of the slab; and shedding to perimeter soils, where local recharge can substantially exceed the regional value. These critical vertical flux components cannot be captured by simply applying a regional recharge rate to the pad footprint. A more nuanced approach is needed, and several promising technologies are discussed in the following sections.

Identifying Focused Recharge with LiDAR and Drone Survey

Newell et al. (2023) proposed that recharge from PFAS source areas is generally diffuse, reasoning that these facilities were not constructed in dry streambeds or depressions, which are often cited as examples of focused recharge. Given the preceding discussion, that conclusion warrants a second look at some sites. Fire training areas are often engineered drainage structures, incorporating graded and crowned pads, berms, curbs, collection sumps, oil–water separators, drainage swales, and containment aprons. Accordingly, at these sites, the potential for focused recharge is more likely driven by small-scale features (i.e., cracks, joints, and interfaces) within engineered structures than by regional topography.

Technological advances over the last decade have made focused-recharge assessment more accessible. One-meter light detection and ranging (LiDAR) is publicly available across much of the United States through the US Geological Survey (USGS) 3D Elevation Program, and drone photogrammetry of a single fire training area can often be completed in a day, yielding centimeter-scale topography. These elevation data can be analyzed with standard flow-routing tools to identify crowned pads, drainage swales, low spots, microcatchments, run-on areas, and other points where runoff may concentrate. The resulting maps provide a practical, physically intuitive basis for identifying candidate focused-recharge zones that warrant field verification or more detailed recharge evaluation (Wilson et al. 2012).

For a more quantitative estimate of potential focused recharge, a simple water balance can be applied to each delineated infiltrating area: precipitation plus runoff from the contributing paved area, minus evapotranspiration, approximates the water available for recharge. As with high-resolution topography, local-scale climate data are increasingly accessible. The publicly available OpenET platform (https://etdata.org/) provides satellite-derived estimates of actual evapotranspiration at 30-m resolution from an ensemble of six remote sensing models served through Google Earth Engine (Melton et al. 2022). Local-scale precipitation can also be obtained from publicly available sources, including:

- NASA (https://gpm.nasa.gov/data)
- NOAA (https://www.ncei.noaa.gov/cdo-web/ and https://psl.noaa.gov/data/gridded/tables/precipitation.html)
- USGS (https://www.usgs.gov/data/precipitation-data)
- Local gauges, gridded climate datasets, and CHIRPS (https://www.chc.ucsb.edu/data/chirps).

Geophysical Methods for Characterizing Subsurface Recharge Pathways

High-resolution topographic data can identify surface features that concentrate runoff and indicate where focused recharge may begin, but these data do not resolve subsurface pathways. Conversely, direct measurements such as borings, lysimeters, and soil-moisture probes provide valuable information at discrete points but are often too costly to deploy at the density needed for local-scale recharge characterization. This leaves important uncertainty in how recharge is distributed below the surface, particularly where preferential pathways or heterogeneous vadose-zone materials control water movement.

Geophysical methods can help fill this scale gap by cost-effectively mapping spatially variable subsurface conditions between direct-measurement locations. Several methods have demonstrated usefulness for identifying potential recharge pathways at refined spatial resolution (Cook et al. 1992; Lawrie et al. 2012; Mawer et al. 2013, 2016; Sendrós et al. 2013; Parsekian et al. 2014; Walsh et al. 2014; Maliva et al. 2015; Gottschalk et al. 2017). Resistivity-based methods are especially useful because contrasts in electrical resistivity commonly correspond to differences in sediment texture, moisture content, or pore-fluid composition. In general, coarse-grained sands and gravels exhibit higher resistivity than clay-rich sediments, although site-specific calibration is needed to support reliable interpretation.

Two promising resistivity-based approaches are tow-behind transient electromagnetic imaging (tTEM) and temporal electrical resistivity imaging (TERI). The tTEM system can be towed behind an all-terrain vehicle to rapidly collect site-scale data and generate three-dimensional models of subsurface electrical resistivity to depths of approximately 60 m (Auken et al. 2019; Behroozmand et al. 2019; Goebel and Knight 2021). This makes tTEM useful for identifying laterally extensive stratigraphic features, coarse-grained intervals, or other potential preferential pathways, including at sites with relatively deep vadose zones.

TERI provides a complementary temporal perspective by collecting repeated apparent-resistivity measurements along fixed electrode arrays. Inversion of these time-lapse datasets can delineate subsurface wetting patterns, macropore structure, and preferential flow behavior (Halihan et al. 2021, 2023; Fields et al. 2022). This temporal component is particularly valuable for recharge characterization because it can help identify not only where recharge occurs but also when recharge fronts move through the vadose zone.

Because resistivity can be influenced by soil moisture, fluid composition, temperature, and particle-size distribution, geophysical results should not be interpreted in isolation. Their value is greatest when calibrated with site-specific information, such as borings, HRSC data, soil samples, lysimeters, or collocated moisture and matric-potential sensors. Used in combination with high-resolution topography and direct measurements, geophysical methods can reduce uncertainty in recharge CSMs by improving characterization of subsurface water movement, preferential pathways, and the spatial and temporal distribution of recharge.

Lysimeters

Specialized hydrologic lysimeters, including pan lysimeters and drain-gauge lysimeters, can provide direct measurements of recharge rates under suitable site conditions. Both designs place a collection device beneath relatively undisturbed soil. Pan lysimeters are installed in an alcove excavated from the face of a vertical trench, whereas drain-gauge lysimeters enclose an intact soil column within a divergence-control tube installed in a borehole. Although these devices have proven useful in some recharge studies, Walker et al. (2026) had limited success using drain-gauge lysimeters to measure recharge directly at two AFFF-impacted sites, largely because of installation artifacts and related problems.

Another type of lysimeter, a zero-tension or gravity lysimeter, can directly estimate recharge by collecting and measuring water infiltrating through a large, relatively undisturbed section of the unsaturated zone at a depth below most root systems. Water that percolates below the root zone is assumed to continue to the water table, providing a direct estimate of recharge volume. However, this method does not address the timing of when this water reaches the water table. Key limitations of this methodology include expense, installation difficulty, point-scale estimates only, and soil disturbance during installation that may affect percolate collection for several years afterward.

Porewater Sampling

Having considered the flow component of the mass flux/mass discharge framework, it is equally important to consider the contaminant concentration component. How can we reliably and cost-effectively characterize the local-scale spatial and temporal variability of contaminant mass in the vadose zone? Suction cup lysimeters, initially developed for measurement of soil moisture content in agricultural settings, have been utilized in several field studies to evaluate PFAS concentrations in porewater and vadose-zone source strength at AFFF-impacted sites

(e.g., Quinnan et al. 2021; Anderson et al. 2022; Schaefer et al. 2022b, 2023). Lysimeter porewater samples can reliably characterize PFAS concentrations across a wide range of soil moisture contents when a minimum sample volume of 10 mL is obtained (Carey et al. 2026). Porewater concentration data can be combined with soil moisture and matric-potential data collected from collocated probes to support high-resolution characterization of vadose-zone PFAS flux to groundwater.

Shallow lysimeters installed at sites where groundwater occurs at less than 3 feet below ground surface increases the risk of mixing with groundwater impacted with PFAS due to the presence of the capillary fringe. For sites located in arid or semi-arid climates (cold or hot) or high-altitude regions, where the depth to groundwater can exceed 100 feet below ground surface, the pathway for PFAS migration from the unsaturated zone to groundwater may be highly complex with a lower probability of leaching PFAS concentrations that reach deep groundwater and a higher probably of surface flow processes (e.g., overland transport and snow/ice) that accumulate water in low areas of ponding, discharge areas, or diversion of stormflow to retention areas distal to the source area (e.g., playa lakes or retention basins) as the primary transport mechanism in these environments (McMillan et al. 2025). In addition, the maximum installation depth for suction lysimeters should generally be limited to less than 25 feet below ground surface due to increasing challenges with proper installation and the risk of reducing porewater sample integrity due to the elevated suction levels (vacuum) required to lift porewater to the ground surface during sampling.

Conventional suction-cup lysimeters for porewater sample collection have been installed in individual vertical or angled borings (e.g., 30° to 45° from vertical); however, these lysimeters can also be installed horizontally (Grossman and Udluft 1991; Lajtha et al. 1999). The "H-Lysimeter" has been proposed by Dr. Chadi Al Motar of the University of Texas at Austin (Figure 9). This concept uses a modified porous-cup suction lysimeter installed within a horizontal boring, such as the VWS, allowing placement of multiple lysimeters within a single horizontal well. The system could be installed beneath pavement, buildings, and other surface infrastructure. It could collect porewater samples from discrete intervals along laterally continuous stratigraphic zones while minimizing disturbance to site conditions and surface infrastructure. This approach would provide much greater spatial coverage than conventional lysimeter installations, which require a separate boring for each sampling point. H-lysimeters and collocated soil-moisture probes could be used at suspected recharge zones (e.g., fractures or joints) beneath concrete slabs where AFFF was historically used to generate high-spatial-resolution PFAS flux data cost-effectively.

Closing Thoughts

To be clear, while we believe that a more focused consideration of site-specific recharge characteristics is needed in general, we are not suggesting that every site requires a comprehensive recharge study. Instead, just as every CSM should, to the extent practicable and consistent with remedial objectives, characterize local-scale geologic heterogeneities (depositional environment, sequence stratigraphy, preferential pathways, etc.), every CSM should explicitly consider how local-scale recharge drives contaminant fate and transport following that same concept. In other words, if the underlying

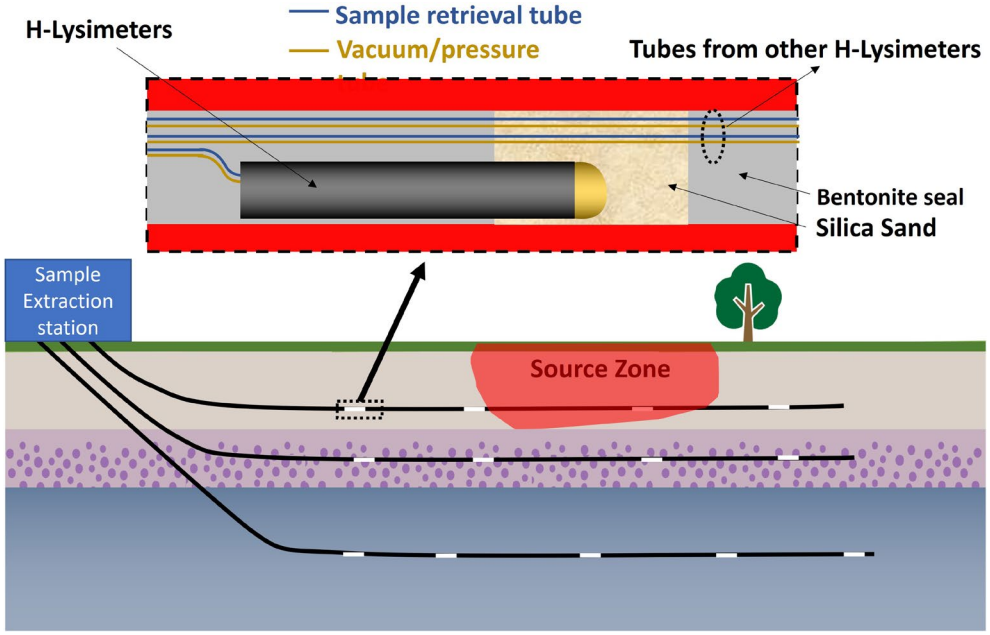


Figure 9. Conceptual schematic of nested H-lysimeters installed within a horizontal well, such as the Vertebrae well system (VWS; Koenigsberg et al. 2018). This proposed system could be installed in the vadose zone, within the capillary fringe, or beneath the water table, including beneath pavement, buildings, and other surface infrastructure where conventional vertical lysimeter borings may be impractical or disruptive. Image courtesy of Chadi Al Motar.

geologic framework serves as the roadmap, then the spatial and temporal distribution of recharge (yes…and discharge [Bredehoeft 2007; Stephens 2009]) provides a kind of "traffic report." And, if our objective is to find and address the mass that is moving, we need to understand this traffic report—where it is backed up, heavy, moving, or free and clear. Putting recharge in this perspective can test assumptions that often go unexamined: whether a source area is truly isolated by "impermeable" surfaces, whether groundwater velocities inferred from hydraulic gradients are temporally stable, whether concentration trends reflect contaminant depletion or recharge-driven dilution and flushing, and whether mass loading is controlled by the highest concentrations or by lower-concentration areas that receive more infiltration.

The data and tools needed for recharge estimation are increasingly available and becoming more inexpensive. High-frequency water levels from pressure transducers, precipitation records, soil moisture or matric-potential sensors, drone- or LiDAR-derived topography, and routine concentration data can provide a low-cost foundation for estimating recharge timing, magnitude, and spatial variability. In many settings, simple methods are entirely appropriate. Empirical recharge estimates, baseflow approaches, or the water-table fluctuation method may be sufficient where the site is relatively homogeneous, recharge is diffuse, the remedial question is not highly sensitive to small differences in flux, and multiple simple approaches converge on a reasonable range.

A closer look at recharge is warranted when recharge uncertainty materially affects decisions about risk, remedy selection, source prioritization, or long-term stewardship. Such scenarios are not exclusive to sites contaminated with PFAS; this approach also applies to sites with other persistent vadose-zone source mass, sites with engineered or paved surface cover and aging underground water infrastructure near release areas, arid and semi-arid sites where recharge is episodic, and sites where plume behavior cannot be reconciled with a steady annual recharge assumption. Finally, like the development of any useful and robust CSM, this process should be iterative, with continual refinement as additional information becomes available, and a multiple-lines-of-evidence approach will increase confidence and reduce uncertainty (e.g., high-resolution hydrographs, sensor networks, porewater sampling, tracer methods, geospatial analysis, numerical modeling, etc.). In this manner, we gain a more refined understanding of recharge and its impact on risk and remedial action.

Recharge is a critical but often under-characterized stimulus of groundwater flow and contaminant mass flux, particularly for per- and polyfluoroalkyl substances (PFAS) and other constituents retained in the vadose zone. Because it controls when, where, and how contaminant mass is mobilized to groundwater, recharge should be evaluated at spatial and temporal scales relevant to contaminated-site decision-making. Improved characterization of recharge can strengthen conceptual site models, support more defensible mass flux and mass discharge estimates, and improve the design and prioritization of remedial strategies. Key practical takeaways from this column are as follows:

- Recharge should be thought of as a dynamic site process, not merely as a static input parameter.
- The most important recharge question is often not "What is the annual recharge rate?" but rather "Where and when does recharge mobilize contaminant mass?"
- PFAS source-zone CSMs should explicitly consider event-scale recharge, focused recharge, and recharge through or around engineered surfaces.
- Cost-effective tools already exist to improve recharge interpretation, including precipitation records, water-level transducers, soil moisture and matric-potential sensors, LiDAR or drone-derived topography, porewater sampling, and simple water-balance methods.
- Recharge characterization should be scaled to the remedial decision: simple approaches are often sufficient, but more detailed evaluation is warranted where recharge uncertainty affects risk, source prioritization, remedy selection, or long-term site management decisions.
- Recharge *redux* simply means intentionally bringing recharge back into appropriate focus and recognizing its fundamental role in contaminant hydrogeology.

Supporting Information

Additional Supporting Information may be found in the online version of this article. Supporting Information is generally not peer reviewed.

Audio S1. Audio Summary.

Table S1. Conceptual Figure Calculations.

References

Aller, L., T. Bennett, J.H. Lehr, R.J. Petty, and G. Hackett. 1985. *DRASTIC: A Standardized System for Evaluating Ground Water Pollution Potential Using Hydrogeologic Settings*. Ada, OK: U.S. Environmental Protection Agency, Office of Research and Development, Robert S. Kerr Environmental Research Laboratory.

Alley, W.M. 1984. On the treatment of evapotranspiration, soil moisture accounting, and aquifer recharge in monthly water balance models. *Water Resources Research* 20, no. 8: 1137–1149.

Allison, G.B., G.W. Gee, and S.W. Tyler. 1994. Vadose-zone techniques for estimating groundwater recharge in arid and semiarid regions. *Soil Science Society of America Journal* 58: 6–14.

Anderson, R.H., J.B. Field, H. Dieffenbach-Carle, O. Elsharnouby, and R.K. Krebs. 2022. Assessment of PFAS in collocated soil and porewater samples at an AFFF-impacted source zone: Field-scale validation of suction lysimeters. *Chemosphere* 308, no. 2022: 136247.

Auken, E., N. Foged, J.J. Larsen, K.V.T. Lassen, P.K. Maurya, S.M. Dath, and T.T. Eiskjær. 2019. tTEM: A towed transient electromagnetic system for detailed 3D imaging of the top 70 m of the subsurface. *Geophysics* 84: E13–E22.

Behroozmand, A.A., E. Auken, and R. Knight. 2019. Assessment of managed aquifer recharge sites using a new geophysical imaging method. *Vadose Zone Journal* 18, no. 1: 1–13.

Bredehoeft, J.D. 2007. It is the discharge. *Ground Water* 45, no. 5: 523.

Bredehoeft, J.D. 1997. Safe yield and the water budget myth. *Ground Water* 35, no. 6: 929.

Bredehoeft, J.D., S.S. Papadopulos, and H.H. Cooper Jr. 1982. Groundwater: The water-budget myth. In *Scientific Basis of*

Water-Resource Management, Studies in Geophysics, 51–57. Washington, DC: National Academy Press.

Brusseau, M.L., and B. Guo. 2023. Revising the EPA dilution-attenuation soil screening model for PFAS. *Journal of Hazardous Materials Letters* 4: 100077.

Brutsaert, W. 1982. *Evaporation into the Atmosphere: Theory, History, and Applications*. Dordrecht: D. Reidel Publishing Company.

Carey, G.R., D.T. Adamson, R.K. Krebs, M. Rebeiro-Tunstall, K.G. Mumford, B. Guo, G.T. Carey, K. Rooney, and S. Moga. 2026. PFAS vadose zone mass discharge at an AFFF-impacted site: 1. Validation based on a multiple lines-of-evidence approach. *Remediation Journal*. https://doi.org/10.1002/rem.70058

Cook, P.G., G.R. Walker, G. Buselli, I. Potts, and A.R. Dodds. 1992. The application of electromagnetic techniques to groundwater recharge investigations. *Journal of Hydrology* 130: 201–229.

Costanza, J., M. Arshadi, L.M. Abriola, and K.D. Pennell. 2019. Accumulation of PFOA and PFOS at the air–water interface. *Environmental Science & Technology Letters* 6, no. 8: 487–491.

Darcy, H. 1856. *Les Fontaines Publiques de la Ville de Dijon*. Paris: Dalmont.

Delin, G.N., and W.N. Herkelrath. 2007. Use of soil moisture probes to estimate ground water recharge at an oil spill site. *Journal of the American Water Resources Association* 41, no. 6: 1259–1277.

Divine, C., K. Hasbrouck, B. Guo, M. Brusseau, J. Zeng, J. Wright, E. Fortner III., S. Chapman, J. Munn, B. Parker, and B. Packer. 2024. Dynamic storage, release, and enrichment of some per- and polyfluoroalkyl substances in the groundwater table fluctuation zone: Transport processes requiring further consideration. *Groundwater Monitoring and Remediation* 44, no. 4: 11–20. https://doi.org/10.1111/gwmr.12694

Douglas, G.B., J.L. Vanderzalm, J.K. Kirby, M. Williams, T.P. Bastow, M. Bauer, K.C. Bowles, D. Skuse, R.S. Kookana, and G.B. Davis. 2023. Sealants and other management strategies for PFAS-contaminated concrete and asphalt. *Current Pollution Reports* 9, no. 4: 603–622. https://doi.org/10.1007/s40726-023-00276-5

Fields, J. Jr., T. Tandy, T. Halihan, R. Ross, D. Beak, R. Neill, and J. Groves. 2022. Electrical resistivity imaging of an enhanced aquifer recharge site. *Journal of Geophysics and Engineering* 19, no. 5: 1095–1110.

Goebel, M., and R. Knight. 2021. Recharge site assessment through the integration of surface geophysics and cone penetrometer testing. *Vadose Zone Journal* 20: e20131.

Gottschalk, I.P., T. Hermans, R. Knight, J. Caers, D.A. Cameron, J. Regnery, and J.E. McCray. 2017. Integrating non-colocated well and geophysical data to capture subsurface heterogeneity at an aquifer recharge and recovery site. *Journal of Hydrology* 555: 407–419.

Grossman, J., and P. Udluft. 1991. The extraction of soil water by the suction-cup method: A review. *Journal of Soil Science* 42, no. 1: 83–93.

Guo, B., J. Zeng, and M.L. Brusseau. 2020. A mathematical model for the release, transport, and retention of per- and polyfluoroalkyl substances (PFAS) in the vadose zone. *Water Resources Research* 56, no. 2: e2019WR026667.

Halihan, T., B.S. Acharya, J.P. Hager, L. Guertault, and G.A. Fox. 2023. Preferential flow velocity mapping of alluvial soil using temporal electrical resistivity imaging. *Discover Water* 3, no. 1: 1.

Halihan, T., J.P. Hager, L. Guertault, and G.A. Fox. 2021. Detecting macropore fingering using temporal electrical resistivity imaging. *Applied Engineering in Agriculture* 37, no. 5: 861–870.

Healy, R.W. 2010. *Estimating Groundwater Recharge*. Cambridge: Cambridge University Press.

Healy, R.W., and P.G. Cook. 2002. Using groundwater levels to estimate recharge. *Hydrogeology Journal* 10: 91–109.

Heyer, B.R., T.C. Osorno, B.A. Carrera, C.M.W. Mok, and J.F. Devlin. 2022. Water flux profiling in fractured rock boreholes with an in-well point velocity probe (IWPVP). *Journal of Hydrology* 613: 128383.

Heyer, B.R., T.C. Osorno, and J.F. Devlin. 2021. Laboratory testing of real-time flux measurements in fractured media. *Journal of Hydrology* 601: 126639.

Kempf, A., G. Leone, and C. Divine. 2013. Field performance of point velocity probes at a tidally influenced site. *Remediation Journal* 23, no. 2: 51–75. https://doi.org/10.1002/rem.21337

Koenigsberg, S.S., E.R. Piatt, and L.I. Robinson. 2018. New perspectives in the use of horizontal wells for assessment and remediation. *Remediation* 28: 45–50. https://doi.org/10.1002/rem.21575

Kolade, S.O., K.E.S. Klint, C. Tsakiroglou, A. Ronen, and O. Dahan. 2026. Dynamics of PFAS transport in the unsaturated zone: Insight from long-term monitoring of an AFFF-contaminated site. *Journal of Hazardous Materials* 516: 143328. https://doi.org/10.1016/j.jhazmat.2026.143328

Labaky, W., J.F. Devlin, and R.W. Gillham. 2009. Field comparison of the point velocity probe with other groundwater velocity measurement methods. *Water Resources Research* 45: W00D30.

Lajtha, K., W. Jarrell, D. Johnson, and P. Sollins. 1999. *Collection of Soil Solution*, 166–182. Oxford: Oxford University Press, Incorporated.

Lawrie, K.C., R.S. Brodie, P. Dillon, K.P. Tan, D. Gibson, J. Magee, J.D.A. Clarke, P. Somerville, L. Gow, L. Halas, H.E. Apps, D. Page, J. Vanderzalm, J. Abraham, S. Hostetler, N.B. Christensen, K. Miotlinski, R.C. Brodie, M. Smith, and G. Schoning. 2012. *Assessment of Conjunctive Water Supply Options to Enhance the Drought Security of Broken Hill, Regional Communities and Industries*. Canberra, ACT: Geoscience Australia.

Lerner, D.N., A.S. Issar, and I. Simmers. 1990. Groundwater recharge: A guide to understanding and estimating natural recharge. In *International Association of Hydrogeologists, International Contributions to Hydrogeology*, Hannover, West Germany: Verlag Heinz Heise. Vol. 8.

Maliva, R.G., R. Herrmann, K. Coulibaly, and W. Guo. 2015. Advanced aquifer characterization for optimization of managed aquifer recharge. *Environmental Earth Sciences* 73: 7759–7767.

Mawer, C., A. Parsekian, A. Pidlisecky, and R. Knight. 2016. Characterizing heterogeneity in infiltration rates during managed aquifer recharge. *Groundwater* 54: 818–829.

Mawer, C., P. Kitanidis, A. Pidlisecky, and R. Knight. 2013. Electrical resistivity for characterization and infiltration monitoring beneath a managed aquifer recharge pond. *Vadose Zone Journal* 12, no. 1: 1–20.

McMillan, H., R. Araki, L. Bolotin, D.H. Kim, G. Coxon, M. Clark, and J. Seibert. 2025. Global patterns in observed hydrologic processes. *Nature Water* 3: 1–10. https://doi.org/10.1038/s44221-025-00407-w

Meinzer, O.E., and N.D. Stearns. 1929. A study of ground water in the Pomperaug Basin, Connecticut, with special reference to intake and discharge. *U. S. Geological Survey Water-Supply Paper* 597-B: 73–146. https://doi.org/10.3133/wsp597B

Melton, F.S., J. Huntington, R. Grimm, J. Herring, M. Hall, D. Rollison, T. Erickson, R. Allen, M. Anderson, J.B. Fisher, A. Kilic, G.B. Senay, J. Volk, C. Hain, L. Johnson, A. Ruhoff, P. Blankenau, M. Bromley, W. Carrara, B. Daudert, C. Doherty, C. Dunkerly, M. Friedrichs, A. Guzman, G. Halverson, J. Hansen, J. Harding, Y. Kang, D. Ketchum, K. Knipper, S. Seiler, W. Kustas, A. Purdy, P. ReVelle, M. Schull, K. Semmens, A. Torres-Rua, K. Wang, Y. Yang, R. Anderson, B. Blake, T. Boucher, J. Deines, K. Gallo, D. Howard, T. Liu, M. McCabe, C. McCraine, T. McVicar, L. Oreopoulos, and J. Volk. 2022. OpenET: Filling a critical data gap in water management for the western United

States. *Journal of the American Water Resources Association* 58, no. 6: 971–994.
Neville, A.M. 2011. *Properties of Concrete*, 5th ed. Essex: Pearson Education Limited.
Newell, C.J., E.B. Stockwell, J. Alanis, D.T. Adamson, K.L. Walker, and R.H. Anderson. 2023. Determining groundwater recharge for quantifying PFAS mass discharge from unsaturated source zones. *Vadose Zone Journal.* 22, no. 4: e20262.
Newell, C.J., L.P. Hopkins, and P.B. Bedient. 1990. A hydrogeologic database for ground-water modeling. *Ground Water* 28, no. 5: 703–714.
Osorno, T.C., J.F. Devlin, and G.C. Bohling. 2022. Geostatistics of the Borden aquifer: High-resolution characterization using direct groundwater velocity measurements. *Water Resources Research* 58: e2020WR029034. https://doi.org/10.1029/2020WR029034
Osorno, T.C., J.F. Devlin, and R. Firdous. 2018. An in-well point velocity probe for the rapid determination of groundwater velocity at the centimeter scale. *Journal of Hydrology* 557: 539–546.
Parsekian, A.D., J. Regnery, A.D. Wing, R. Knight, and J.E. Drewes. 2014. Geophysical and hydrochemical identification of flow paths with implications for water quality at an ARR site. *Groundwater Monitoring & Remediation* 34: 105–116.
Quinnan, J., M. Rossi, P. Curry, M. Lupo, M. Miller, H. Korb, C. Orth, and K. Hasbrouck. 2021. Application of PFAS-mobile lab to support adaptive characterization and flux-based conceptual site models at AFFF releases. *Remediation Journal* 31: 7–26.
Reitz, M., W.E. Sanford, G.B. Senay, and J. Cazenas. 2017. Annual estimates of recharge, quick-flow runoff, and evapotranspiration for the contiguous U.S. using empirical regression equations. *JAWRA Journal of the American Water Resources Association* 53, no. 4: 961–983. https://doi.org/10.1111/1752-1688.12546
Ronan, A.D., D.E. Prudic, C.E. Thodal, and J. Constantz. 1998. Field study and simulation of diurnal temperature effects on infiltration and variably saturated flow beneath an ephemeral stream. *Water Resources Research* 34, no. 9: 2137–2153.
Russell, R., B. Guo, J. Zeng, M.L. Brusseau, C.E. Schaefer, S. Shea, C.P. Higgins, and T.P.A. Ferre. 2025. Combining field datasets and mathematical modeling to quantify PFAS leaching and mass discharge at an AFFF-impacted site. *Water Research* 286: 124063.
Salt, G., and T.R. Kjeldsen. 2019. Evaluating the potential for infiltration through urban paved surfaces. *Hydrological Sciences Journal* 64, no. 10: 1237–1249.
Sanford, W. 2002. Recharge and groundwater models: An overview. *Hydrogeology Journal* 10, no. 1: 110–120. https://doi.org/10.1007/s10040-001-0173-5
Scanlon, B.R., R.W. Healy, and P.G. Cook. 2002. Choosing appropriate techniques for quantifying groundwater recharge. *Hydrogeology Journal* 10, no. 1: 18–39. https://doi.org/10.1007/s10040-001-0176-2
Schaefer, C., G. Lavorgna, D. Lippincott, D. Nguyen, J. Schaum, C. Higgins, and J. Field. 2023. Leaching of perfluoroalkyl acids during unsaturated zone flushing at a field site impacted with aqueous film forming foam. *Environmental Science & Technology* 57: 1940–1948.
Schaefer, C.E., D. Nguyen, E. Christie, S. Shea, C.P. Higgins, and J.A. Field. 2022a. Desorption isotherms for poly- and perfluoroalkyl substances in soil collected from an aqueous film-forming foam source area. *Journal of Environmental Engineering* 148, no. 1: 04021074.
Schaefer, C., G. Lavorgna, D. Lippincott, D. Nguyen, E. Christie, S. Shea, S. O'Hare, M. Lemes, C. Higgins, and J. Field. 2022b. A field study to assess the role of air-water interfacial sorption on PFAS leaching in an AFFF source area. *Journal of Contaminated Hydrology* 248: 104001.
Schaefer, C.E., D. Nguyen, E. Christie, S. Shea, C.P. Higgins, and J.A. Field. 2021. Desorption of poly- and perfluoroalkyl substances from soil historically impacted with aqueous film-forming foam. *Journal of Environmental Engineering* 147: 06020006.
Schaefer, C.E., V. Culina, D. Nguyen, and J.A. Field. 2019. Update of poly- and perfluorinated substances at the air-water interface. *Environmental Science & Technology* 53, no. 21: 12442–12448.
Sendrós, A., M. Himi, R. Lovera, I. Casado, and A. Casas. 2013. Estimating recharge potential and preferential flow paths in a meander close to a recharge pond using ERT. *Near Surface Geoscience* 2013: 9–11.
Shah, N., M. Nachabe, and M. Ross. 2007. Extinction depth and evapotranspiration from ground water under selected land covers. *Groundwater* 45, no. 3: 329–338.
Sharifan, H., M. Bagheri, E. Wang, J.G. Burken, C.P. Higgins, Y. Liang, J. Liu, C.E. Schaefer, and J. Blotevogel. 2021. Fate and transport of per-and polyfluoroalkyl substances (PFASs) in the vadose zone. *Science of the Total Environment* 771: 145427.
Slichter, C.S. 1899. *Theoretical Investigations of the Motion of Ground waters*. U.S. Geological Survey 19th Annual Report, Part II, Washington, DC: Government Printing Office. 295–384.
Stephens, D.B. 2009. *Vadose Zone Hydrology*. Boca Raton, FL: CRC Press.
Stephens, D.B., K.C. Hsu, M.A. Prieksat, M.D. Ankeny, N. Blandford, T.L. Roth, J.A. Kelsey, and J.R. Whitworth. 1985. A comparison of estimated and calculated effective porosity. *Water Resources Research* 21, no. 1: 25–32.
Stults, J.F., C.E. Schaefer, Y. Fang, J. Devon, D. Nguyen, I. Real, S. Hoa, and J.L. Guelfo. 2024. Air-water interfacial collapse and rate-limited solid desorption control perfluoroalkyl acid leaching from the vadose zone. *Journal of Contaminant Hydrology* 265: 104382.
Theis, C.V. 1940. The source of water derived from wells. *Civil Engineering* 10, no. 5: 277–280.
Tóth, J. 1962. A theory of groundwater motion in small drainage basins in central Alberta, Canada. *Journal of Geophysical Research* 67, no. 11: 4375–4388. https://doi.org/10.1029/JZ067i011p04375
Vahedian, F., J.A.K. Silva, J. Simunek, and J.E. McCray. 2024. Influence of kinetic air-water interfacial partitioning on unsaturated transport of PFAS in sandy soils. *Science of the Total Environment* 957: 177420.
Walker, K.L.J., C.J. Newell, B. Actkinson, L. Farnell, E. Stockwell, R. Craycroft, H. Stroo, J. Blotevogel, and T. Sale. 2026. *Using Real-Time Remote Sensors to Reduce the Cost of Long-Term Monitoring and Remediation Performance Monitoring at PFAS Vadose Zone Sources (ESTCP Project ER22-7381, Final Report)*. Environmental Security Technology Certification Program. https://serdp-estcp.mil/projects/details/d2ff9af0-237d-4239-a080-94383c8cf7b3/using-real-time-remote-sensors-to-reduce-the-cost-of-long-term-monitoring-and-remediation-performance-monitoring-at-pfas-vadose-source-zones. (accessed September 26, 2026).
Wallis, I., J. Hutson, G. Davis, R. Kookana, J. Rayner, and H. Prommer. 2022. Model-based identification of vadose zone controls on PFAS mobility under semi-arid climate conditions. *Water Research* 225: 119096.
Walsh, D.O., E.D. Grunewald, P. Turner, A. Hinnell, and T.P.A. Ferre. 2014. Surface NMR instrumentation and methods for detecting and characterizing water in the vadose zone. *Near Surface Geophysics* 12: 271–284.
Wiles, T.J., and J.M. Sharp Jr. 2008. The secondary permeability of impervious cover. *Environmental & Engineering Geoscience* 14, no. 4: 251–265.

Wilson, J.P., and J.C. Gallant. 2000. Digital terrain analysis. In *Terrain Analysis: Principles and Applications*, ed. J.P. Wilson, and J.C. Gallant. New York, NY: John Wiley & Sons.

Worthington, S.R.H. 2026. Explaining high celerities and velocities in vadose zone fractures. *Groundwater* 64: 253–257. https://doi.org/10.1111/gwat.70064

Zeng, J., and B. Guo. 2021. Multidimensional simulation of PFAS transport and leaching in the vadose zone: Impact of surfactant-induced flow and subsurface heterogeneities. *Advances in Water Resources* 155: 104015.

Biographical Sketches

***Craig Divine**, corresponding author, Ph.D., P.G., is a Senior Vice President and Principal Hydrogeologist at Arcadis, 8725 Rosehill, Suite 350, Lenexa, KS 66215; (720) 308-5367; craig.divine@arcadis.com*

***Charles Newell**, Ph.D., is a Vice-President and Principal Engineer at GSI Environmental Inc., 2211 Norfolk Street, Ste. 1000, Houston, TX, 77098; (713) 522-6300.*

***Kenneth Walker**, is a Senior Geologist and Engineer at GSI Environmental Inc., 2211 Norfolk Street, Ste. 1000, Houston, TX, 77098; (713) 522-6300.*

***Matthew Spurlin**, P.G., is a Senior Hydrogeologist at Noblis, Inc., 2002 Edmund Hillary Dr., Reston, VA 20191; (703) 610-2000.*

***Timothy Blount**, P.E., is a Project Environmental Engineer at Arcadis, 7550 Teague Road, Suite 310, Hanover, MD 21076.*

***Prashanth Khambhammettu**, Ph.D., P.E., is a Principal Engineer and Groundwater Modeling Practice Area Lead at Arcadis, 7550 Teague Road, Hanover, MD 21076; (410) 923-7823.*

***Marc Killingstad**, P.E., is a Technical Expert, Director of the Hydrogeology Community of Practice, and Vice President at Arcadis, 7550 Teague Road, Hanover, MD 21076; (410) 923-7816.*