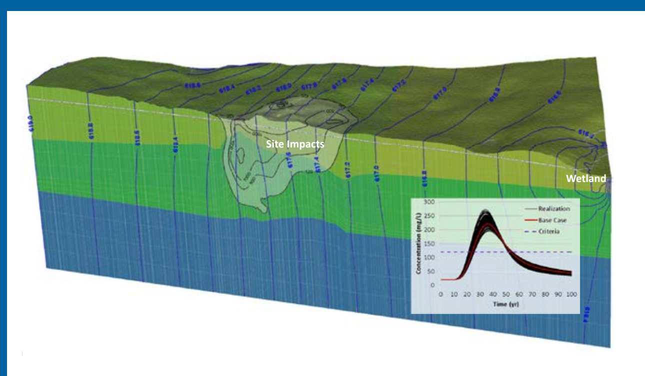
Groundwater: State of the Science and Practice



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Foreword

The National Ground Water Association is a community of professionals working together to advance groundwater knowledge. This text is part of NGWA's goal to bring sound science to practical applications. It presents short articles on groundwater topics of broad appeal across all segments of the groundwater industry.

Experts in each of the subjects covered were solicited to write a short article on the state of the science and practice of their topic—everything from geophysics and groundwater modeling to well design and well maintenance. Each author was given latitude to focus on specific aspects of their choice within their broader topic.

We appreciate the thoughtful and timely way in which all authors contributed to this effort and hope you benefit from their contributions.

Dr. William M. Alley
Director of Science and Technology
National Ground Water Association

Bacteriology of the Water Well

By John H. Schnieders, Ph.D., CPC

With our understanding of microbial communities and microbiomes expanding almost daily, we still, as a society, have difficulty accepting bacteria are universally present in our aquifer systems. Research has shown groundwater wells are dynamic systems, with a wide variety of differences depending on their design, construction, operation, and maintenance, as well as the source aquifers. These differences can result in a variety of conditions downhole which can impact microbial communities and the use of the well and produced water.

Bacteria as They Relate to Drinking Water Safety

Despite the changes to the Total Coliform Rule occurring in 2016 moving coliforms to more of an indicator organism, in the water industry the total coliform test has become the default means of assessing microbial water quality. Unfortunately, and to much consternation, the total coliform test does little to identify the pathogens which are opportune residents of the well, or the well's natural microbial population and their implications on well operation.

Pathogens are any agent that can cause disease and for this chapter are pathogenic bacteria that can cause a disease. They are not normal or usual residents of a well. As occasional residents or transient populations, they are part of the total makeup of the bacteria within the well but share the space with the much larger population of the total well biome.

Understanding the population size and their association is necessary to comprehend the full nature of the bacteria present in our well systems and aquifers. Well bacteria which are the cause for well fouling and aesthetic issues (the natural well biome residents) are also responsible for or make it possible for the presence of the pathogens. Together they make up the total population of bacteria inhabiting our wells and aquifers.

There are two aspects that need to be addressed regarding bacterial involvement in water wells, which are important for very different reasons. Of most importance to the general public and health professional are the pathogenic bacteria accounting for the outbreaks of illness, and the more common occurrence resulting in severe illness in families of well owners. Illness for the small well owner can be caused by bacteria that are known pathogens; however, many more incidences involve opportunistic pathogens, bacteria which are often natural inhabitants of the well or aquifer. Exposure to large populations of bacteria or even smaller exposure if the person is compromised due to illness or age may result in a health issue.

Although there can be severe outbreaks in the public sector, it is more common to see episodes of bacterial infection in the private sector. This is a direct result of the fact municipal systems are better monitored and maintained by law, and typically produce a much higher volume of water on a more regular basis. Research continues to show how active wells are typically healthier than stagnant or partially idled wells. Home or small community wells are often less maintained, produce a smaller volume of water, and often create a concentrated effect on the water users.

Bacteria as They Relate to Well Maintenance

Directly or indirectly, bacteria account for a significant need for maintenance in water wells. The resulting fouling is so significant that maintenance and rehabilitation can account for a very large portion of the cost of well ownership. In municipal and high production wells, this cost often exceeds five times the initial cost of the well when evaluated over the life of the well. This doesn't take into consideration loss of time and water needs. Of course, pathogens also have requirements for maintenance such as disinfection and remediation

Pathogens are often considered singularly; however, they are dependent on large populations of other bacteria which create and maintain the environment necessary for the pathogens' entry into and survival within the well environment. This coexistence, and in many cases dependency, predicates the need for well maintenance.

Not only does the well suffer from well plugging and often aesthetic water quality issues, such as turbidity and taste, but allowing buildup of large non-pathogen populations within a well severely compromises the well's ability to self-clean. Nuisance bacteria as they are often called can make up huge populations within the well environment providing shelter, food, and the specific chemical biome for the pathogen to set up residency and survive to contaminate water production. Essentially, a dirty well is subject to or at a greater risk for health issues than a properly maintained water well.

These nuisance bacteria cause severe damage to water quality and production while providing the ideal environment for pathogenic organisms. They consist of several groups of bacteria which are loosely referred to by their most obvious characteristics. While each group has its own detrimental effect on the well, it is difficult to classify one more detrimental than the other. Usually the well suffers from several different groups; however, it would be possible for all to be present—especially in a poorly maintained system.

Iron bacteria are probably the best known. These organisms are so classified due to their ability to oxidize iron. They essentially make the iron soluble in the water—causing discoloration, fouling, and taste problems. The most commonly identified species is Gallionella; however, many aerobic organisms are capable of oxidizing iron. An iron bacteria occurrence can impact multiple aspects of the well including produced water quality, treatability or use of the produced water, and operation of the pump and well system. Their occurrence in the well requires different methods of cleaning.

Slime formers are another common group. In some respect, all bacteria produce slime, a polysaccharide exopolymer or biofilm, as a means of nutrient capture and protection. Slime formers are a group of primarily aerobic organisms that produce an excessive amount of slime, in some cases as much as 1000 times their own cell mass. Slime formers often reside in the near-well formation, throughout the gravel pack, and within the well proper above the anaerobic zone, thus impacting the most active portions of the well.

In addition to the development of slime and the direct impact it can have on flow, these formations can act as a point of accumulation for precipitating mineral assemblages, migrating sediment, and other bacteria. Within the well column they are the largest user of oxygen and in doing so promote corrosion through by-product production. Their death due to oxygen starvation results in the accumulation of considerable organic debris in the well bottom. Their anoxic death occurs when the well is not pumped, resulting in no fresh water being brought into the well so that it is repeated each well cycle, usually daily.

Anaerobic bacteria, while often less disseminated than aerobic populations, are another important group to consider. Their largest populations are usually found in the well bottom but may extend to the gravel pack or karst formations near the well. The well bottom is often devoid or depleted of oxygen resultant from hydrologic isolation, lower producing zones, or outright isolation due to use of blank casing or poor development.

As part of their normal activity, anaerobic bacteria produce a particularly dense form of biofilm to limit oxygen. Anaerobic bacteria also produce considerable acid in their confined environment, creating a zone of low or acidic pH. Anaerobic environments nurture growth of certain pathogens or opportunistic pathogens that might enter the zone who otherwise would pass through the well without multiplying their numbers. Since better than 90% of known pathogens are anaerobic, this area becomes their primary incubation area for contamination. Coliforms specifically are facultative anaerobes, generally preferring the anaerobic zone of the well which provides more nutrients as well as protection from harmful interactions, including simple chlorination.

The bacteria making up one of the largest populations of anaerobic organisms found in the lower extensions of the well are the sulfate-reducing bacteria. These are the bacteria which are most responsible for the characteristics of this zone. They reduce sulfur, thereby producing acidic conditions and hydrogen sulfide (H_2S) gas. This can result in severe corrosion of the well structure as well as the distinct rotten egg odor at startup of the well. The H_2S gas is easily dissolved in water and can be an active source of corrosion in an idle well, impacting the upper well structure and even associated piping and storage systems.

Every well professional is familiar with the thick black foul-smelling material that is taken from the well following airlifting of the debris from the well bottom. The black color is iron sulfide formed by many of the anaerobic bacteria. Anaerobic growth is an excellent indicator of the extent of microbial presence and maturity, and as such may reflect impacted flow or other influence on the well and near-well aquifer interface. As an assessment of produced water quality, monitoring of the anaerobic population is important as many coliforms and opportunistic pathogens are anaerobic or facultative anaerobes.

Assessment of Total Bacterial Populations

Monitoring the total microbial load or population is an excellent means of quantifying the bacterial presence.

In the past decade, many industries that deal with microbial quality have embraced the adenosine triphosphate (ATP) method of assessing the total microbial population present in an environment. Previously, many relied on heterotrophic plate counts as a means of quantifying the microbial load; however, these methods have and continue to rely on growth agar to which many bacteria don't respond. The ATP method allows for a more accurate and expedited means of quantifying the total viable bacterial population present in a water sample or on a surface, regardless of the type of bacteria.

Monitoring the fluctuations in the microbial community is a valuable means of identifying an increasing bacterial load which could indicate an increased potential for nuisance and problematic organisms.

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Carbon Sequestration

By Michael Celia and Ryan Edwards

Overview of Geological Carbon Storage

The technology known as carbon capture and storage[1], or CCS, involves capture of carbon dioxide emissions from large stationary sources and subsequent injection of the captured CO_2 into suitable deep geological formations. The subsurface storage part of the operation is often referred to as Geological Carbon Storage, or GCS. A suitable storage formation is one that is deep enough for the injected CO_2 to be in a supercritical state, large enough to store significant amounts of CO_2, and permeable enough for injection rates to match capture rates. The formation also needs to be overlain by a low-permeability caprock formation that will keep the CO_2 in the injection formation over time scales from centuries to millennia.

Target formations include depleted oil and gas reservoirs and deep saline aquifers. Injection into depleted oil reservoirs is usually associated with enhanced oil recovery (EOR) while injection into deep saline aquifers is for dedicated storage. EOR is a well-established technology with potential demand for significant amounts of CO_2. But the largest storage capacity, by far, resides in deep saline aquifers, which have the added advantage of wide global distribution (IPCC 2005; North American Carbon Storage Atlas 2012; Szulczewski et al. 2012).

Carbon dioxide is in a supercritical state when the pressure exceeds 7.4 mega-Pascal (MPa) and the temperature exceeds 31 degrees Celsius (°C). With density of supercritical CO_2 being on the order of 500 kg/m^3, the stored CO_2 is much denser than gaseous CO_2, thereby providing greatly improved volumetric storage efficiency, but is still much lighter than the background brine in a typical deep saline formation. The density difference between CO_2 and brine leads to strong buoyancy in the system, hence the need for a competent overlying caprock. The injected CO_2 is only slightly soluble with the brine, so that most of the CO_2 remains in a separate fluid phase, leading to a two-phase flow problem. For EOR, the phase behavior and resulting fluid dynamics can be more complicated (Lake 1989; Sobers et al. 2013; Ettehadtavakkol et al. 2014), and in all cases, computational modeling as well as system monitoring pose a number of challenges (Celia et al. 2015; Jenkins et al. 2015).

It is useful to distinguish two different kinds of sources for CO_2 capture: those sources with an essentially pure stream of CO_2, like ethanol plants, and those with a dilute stream, like traditional power plants. Pure streams of CO_2 are inexpensive to capture, and form the sources for most of the existing CCS operations around the world (Global CCS Institute 2018). Unfortunately, these concentrated streams form only a small fraction of the overall CO_2 emissions from stationary sources. Capture of CO_2 from dilute streams is still quite expensive, and this high capture cost helps to explain the lack of a large-scale CCS industry.

Almost all analyses of pathways to limit average global warming to 2°C by 2100, as specified in the Paris Agreement of 2015 (UNFCCC 2015), include massive amounts of CCS, on the order of gigatonnes (Gt) of CO_2 per year (Rockstrom et al. 2017). This includes "traditional" CCS associated with capture of emissions from fossil fuel combustion as well as so-called Bio-Energy with CCS, or BECCS, where the combustion fuel is biological in origin such as trees or crop residues (Kemper 2015). Future projections

[1] In this chapter we use the term "storage" instead of "sequestration." In the context of CCS, the two words tend to be used interchangeably.

usually consider BECCS as a "negative emissions" technology, where the net effect is removal of CO_2 from the atmosphere (Minx et al. 2018; Searchinger et al. 2017); most projections require massive

While there remains an expectation of a massive CCS effort, to date only a few small-scale dedicated storage projects have been developed, and while there are more EOR projects, the total number of projects is fewer than 50 and the total injection rates are 30 to 40 Mt CO_2/yr (Global CCS Institute 2018). The scale-up to injections of 1 to 10 Gt CO_2/yr is a daunting challenge.

The Path to Gigatonne CCS Scale-Up

Appropriate government policies, including a price on carbon emissions, will be necessary for the development of a large-scale CCS industry. While uneven, there has been movement forward in some parts of the world. An example is the development of a nationwide price on carbon emissions in China, which is the world's largest greenhouse gas emitter (Chemnick 2017; Le Quere et al. 2017). This national carbon pricing system is an important development that could eventually influence a large fraction of the nations of the world, especially those in China's Belt and Road Initiative (BRI 2018), which involves roughly half of the world's population.

While the economics and politics of carbon emission pricing are developing, it is important that research on CCS continues, and that opportunities to develop large-scale infrastructure are pursued vigorously. Three specific areas are worth highlighting: (1) environmental risk analysis associated with CCS/GCS; (2) new technologies for carbon capture; and (3) new opportunities to develop large-scale infrastructure needed for CCS scale-up.

(1) *Environmental Risk Analysis:* There are two broad concerns associated with large-scale subsurface injection systems: (i) Potential leakage of fluids, including both the injected CO_2 and brine, out of the injection formation to locations of concern. This includes leakage into shallow drinking water aquifers or to the atmosphere. (ii) Potential induced seismicity. Research on both of these topics appears to indicate that the risks are manageable.

In terms of leakage, there is a general consensus that leakage along concentrated pathways (wells, faults/fractures) represents the greatest risks, with old wells being a particular concern, especially in North America (IPCC 2005; Brandt et al. 2014; Pawar et al. 2015). A combination of modeling and measurements indicates that leakage along old wells is unlikely to be a major problem. Examples include the modeling analysis of Nogues et al. (2012) and the measurements of Crow et al. (2010), Kang et al. (2015), and Tao and Bryant (2014), which in combination appear to predict acceptably small amounts of leakage through old wells.

Earlier concerns about induced seismicity (Zoback and Gorelick 2012) have been countered by more recent studies showing that as long as the pressure increase associated with large-scale CO_2 injection is isolated from the basement rocks of a sedimentary basin, there is low probability of seismic events felt at the land surface (Zhang et al. 2013; Vilarrasa and Carrera 2015). Research on both of these topics needs to continue, so that the confidence level for safe and secure large-scale injections is high when a CCS industry eventually develops.

(2) *New Capture Options:* CO_2 capture costs associated with traditional fossil fuel-based power plants are high because CO_2 is only a small fraction of the exhaust stream. Capture from this dilute stream dominates the cost of the CCS operation, making the economics challenging and leading to the lack of large-scale CCS projects.

While a range of research on capture technologies continues to be pursued, one particularly notable development is a new power generation system driven by fossil fuels but using the so-called Allam cycle (Allam et al. 2014). In this technology, CO_2 is used as the working fluid to drive the turbine in the power system, and an oxy-combustion process leads to an exhaust stream that is essentially pure CO_2. This leads to a low-cost capture opportunity, thereby greatly enhancing the possibility of economically viable CCS applied to the power sector, whose emissions dominate total emissions from stationary sources.

A new pilot plant is in the final stages of development by Net Power LLC (Allam et al. 2017; Net Power 2018), with the company claiming that the power production will be cost competitive with traditional fossil fuel-based power plants while producing a pure stream of CO_2. If this technology proves to be successful, it could radically change the outlook for a large-scale CCS industry.

(3) *Large-Scale Infrastructure Development:* The 2018 U.S. tax bill includes expanded tax credits for CCS (Heitkamp 2017). The so-called 45Q tax credits provide credits of up to 35 U.S. Dollars (USD) per tonne of captured CO_2 when the CO_2 is used for EOR and up to 50 USD/tonne CO_2 for dedicated storage. When coupled with the price of CO_2 that oil companies pay to purchase CO_2 for EOR operations, there appears to be sufficient economic incentives to enable the construction of a large-scale pipeline in the central part of the United States (Edwards and Celia 2018). This pipeline would connect low-capture-cost sources in the Upper Midwest, mostly associated with ethanol plants, with the Permian Basin in west Texas, which is a large demand center for CO_2 EOR. With foresight in the design of the system, the pipeline could be oversized to allow for a much larger set of future capture sources that would include power plants and other large industrial point sources. Figure 1 shows the low-capture-cost sources in the U.S., while Figure 2 shows a viable network to connect low-cost ethanol sources to the Permian Basin. This is an example of how opportunities need to be pursued aggressively when they arise, including considerations for future scale-up, so that when the economics of large-scale CCS become favorable, there are minimal barriers to the rapid development of the industry.

Figure 1. Low-capture-cost CO_2 sources, existing CO_2 pipelines, and deep saline aquifers. Total low-capture-cost emissions are 87 Mt CO_2 per year. Figure from Edwards and Celia (2018).

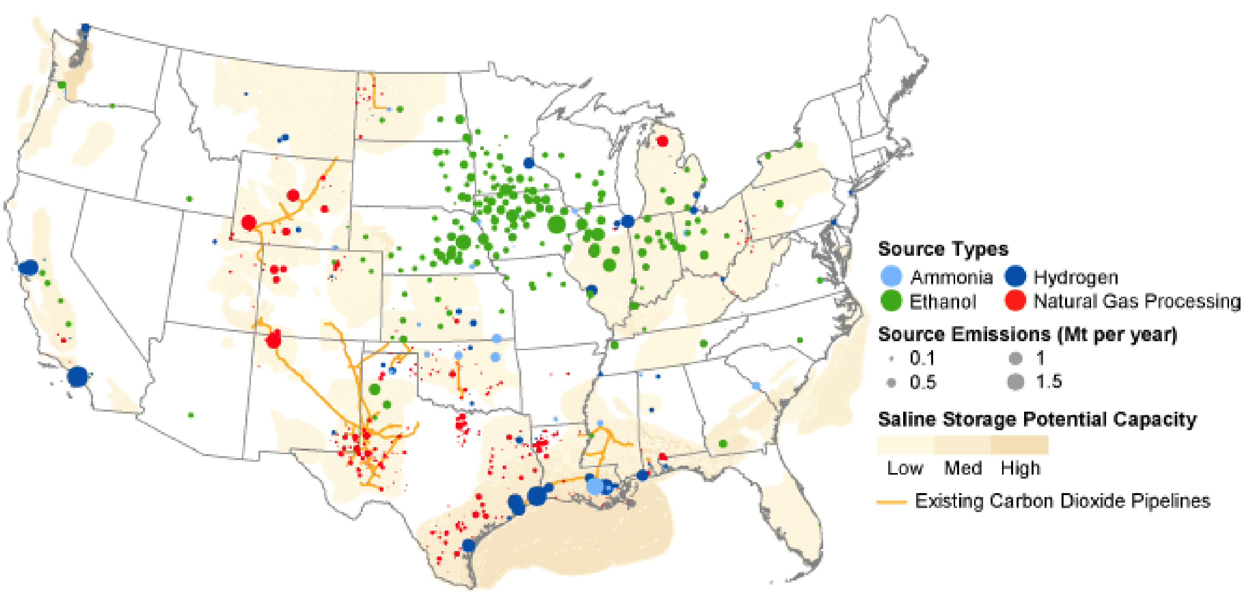
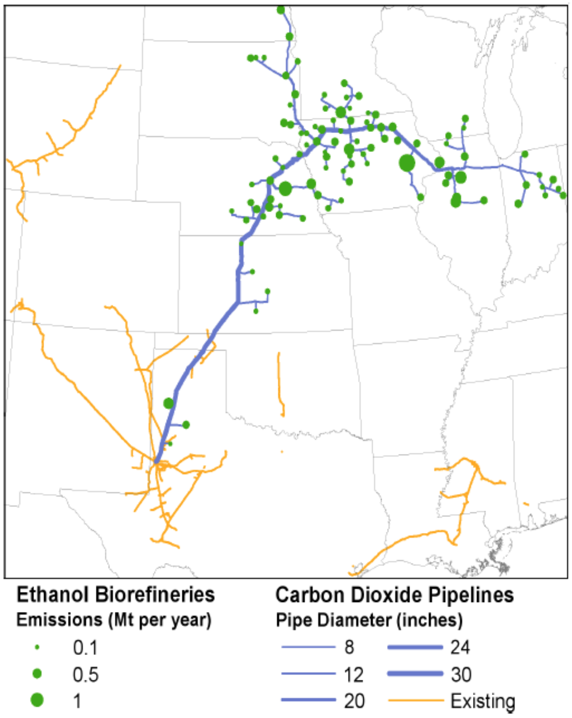


Figure 2. Cost-effective pipeline network capturing emissions from Midwest ethanol plants and delivering it to the EOR locations in the Permian Basin, West Texas. Figure from Edwards and Celia (2018).



Conclusions

Carbon capture and storage (CCS) is the only available technology that allows for continued use of fossil fuels while also addressing the carbon problem. CCS features prominently in most projections of future low-carbon energy systems that achieve the Paris Agreement target for global warming. The enormous scale of these future projections for CCS is inconsistent with the current pace of CCS development, which remains slow. While broad economic and policy incentives are likely to be needed to stimulate a large CCS industry, steps can and should be taken now to prepare for large-scale CCS deployment. These include continued scientific and engineering studies for both capture and large-scale storage, and aggressive movement to build large-scale infrastructure whenever and wherever possible.

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Contaminant Hydrogeology

By Kevin D. Svitana, Ph.D., PG, CP

Introduction

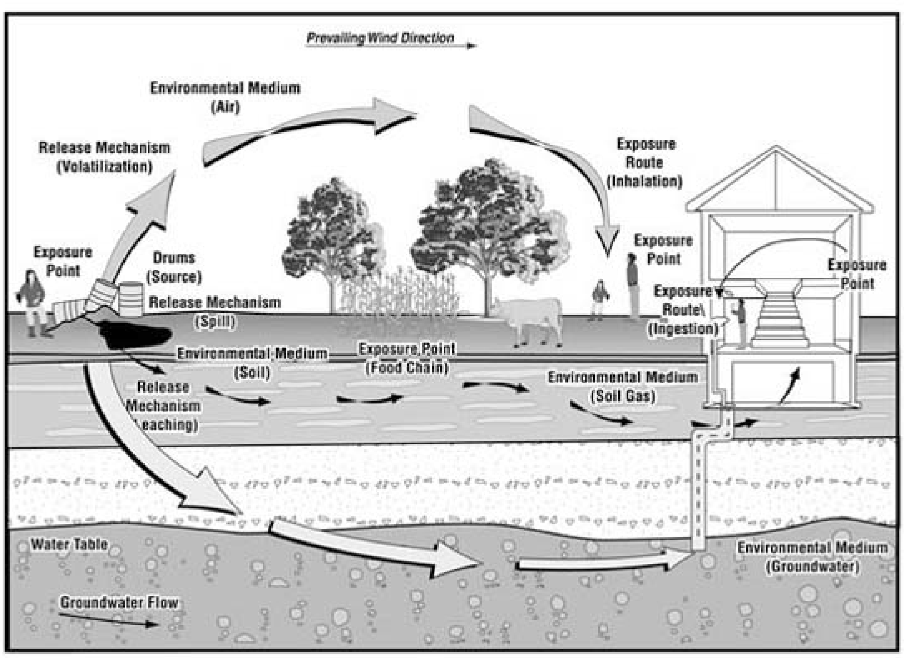
Those who have been practicing in the field of contaminant hydrogeology since the inception of the various United States hazardous waste laws (Resource Conservation and Recovery Act, RCRA; Comprehensive Environmental Response Compensation and Liability Act, CERCLA; or Superfund; etc.) have experienced dramatic changes in technology. The changes have led to improved capabilities toward understanding groundwater/aquifer interactions and contaminant behavior. Assessing the potential fate and transport of contaminants in the subsurface is key to evaluating receptor risk so mitigation efforts can be effectively developed.

Evaluating potential impacts to human health and the environment was outlined by the Agency for Toxic Substances and Disease Registry (ATSDR) in their *Public Health Assessment Guidance Manual* in 1992 to help direct the assessment process. To facilitate communication of contaminant migration pathways to the public, ATSDR added an illustration of a generalized site conceptual model (labeled Figure 6-2 below) to the 2005 revision of the 1992 document.

Regulatory agencies have since put considerable emphasis on developing conceptual site models (CSMs) that are generalized and easy for the public to understand and interpret. However, from the perspective of a contaminant hydrogeologist, emphasis on refining the CSMs and projecting contaminant fate, transport pathways, and receptor risk has been revolutionized by technology advancements over the last 30-plus years.

Several innovations and challenges related to better assessing contaminant hydrology and refining CSMs discussed in this chapter include: improving laboratory and field analytical capabilities; develop-

Figure 6.2: The conceptual site model from the 2005 ATSDR *Public Health Assessment Guidance Manual.*



ing new subsurface exploration and assessment techniques; looking holistically at groundwater systems including the biological effects to contaminants; understanding vapor phase migration; and the continuing challenge presented by emerging contaminants.

Advances in Analytical Capabilities

When the referenced federal hazardous waste programs began, the analytical capabilities of laboratories performing soil and water analysis were often challenged to meet the regulatory minimum detection limits, which typically were concentrations in parts per million (milligrams per liter, 10^{-6}). However, as studies of the toxicity of compounds and analytes like chlorinated solvents, mercury, and dioxins developed, it became apparent that the risk to human health and the environment occurs at concentrations below the parts per billion level (micrograms per liter, 10^{-9}) and, as in the case of dioxin, risk occurs at concentrations in parts per quadrillion (picograms per liter, 10^{-15}, USEPA 2018a).

The ability to consistently meet such low detection limits was a challenge for the analytical laboratories, but as detection limits fell, it became even more important to collect representative samples to avoid matrix interference, cross contamination, and other sampling-induced problems that would invalidate data (Murphy and Morrison 2007).

Regulators and scientists emphasized the need to collect representative samples, so monitoring well constructions were scrutinized to assure the appropriate intervals were sampled. Sampling techniques like three-volume borehole purging gave way to low flow methods and membrane bag sampling to reduce the potential for interferences from sediment entrainment (USEPA 2018). Now the sampling technique is deemed as important as analytical precision.

Field Analytical Methods

There have been corresponding advances with in-the-field analytical capabilities. The improvement in field gas chromatography equipment has moved field analysis of compounds and analytes from being only a screening tool to providing analytical results that can be used as confirmation samples for delineating identified areas (Murphy and Morrison 2007).

Other techniques like membrane interface probes (MIPs) allow for accurate screening for the presence of volatiles in both saturated and unsaturated intervals with resolution thicknesses of centimeters. This direct push technology tool can help identify isolated intervals that could be a significant source area that may have been overlooked using conventional hollow stem auger drilling and soil screening methods. The analytical capabilities of the probe can be enhanced with a hydraulic profiling tool to enable logging of relative formation permeability (McCall 2014). Therefore, both aquifer chemistry and hydraulic characteristics can be accurately assessed on the fly in the field, allowing for real-time assessment and decision-making.

Like MIPs, new methods to quantify transmissivities have led to new approaches for scanning transmissive features within the borehole. The technique described in Keller et al. (2014) uses a water-filled flexible membrane that everts the liner down the borehole, so the liner pushes out borehole water into the transmissive fractures or other permeable features. When compared to conventional packer interval testing to assess fracture flow, using the flexible membrane eliminates the need to manage investigation derived waste (the contaminated groundwater in the borehole) which simplifies project logistics and reduces waste disposal costs. This approach to defining fracture locations and quantifying the hydraulic conductivities of fractures or permeable zones can be a valuable tool for better understanding fractured rock systems.

A persistent problem in fractured rock terrain or areas where there are intervals of contrasting hydraulic conductivities in unconsolidated aquifers is the occurrence of matrix diffusion rebound when groundwater recovery systems are shut off (Steimle 2002). Contaminant concentrations are diluted by the accelerated flow through fractures in highly transmissive zones, and when pumping stops and flow rates through the transmissive zones are reduced, concentrations rebound. The effects of matrix diffusion into fractures and projecting the potential for natural attenuation from microbial degradation is an important consideration when trying to project if concentrations have been reduced sufficiently to facilitate natural attenuation from microbial degradation (Pierce et al. 2018). Contaminant concentration rebound in fractured rock aquifers where only pump and treat remediation has been employed has resulted in remedial actions extending for decades.

Geomicrobiology and Environmental Remediation

Persistent contaminants with long half-lives, like chlorinated solvents, are a challenge to remediate. However, the combined disciplines of hydrogeology and microbiology resulted in innovative approaches to remediate persistent compounds.

A specialization in the geosciences, geomicrobiology is being utilized in the practice of groundwater remediation. The specialization as described by Hernandez-Machado and Casillas-Martinez (2009) is a blend of biology, microbiology, and Earth sciences and focuses on the skills to evaluate microbial populations in soil and bedrock environments. The application of microbial assessment to in situ remediation is further described by Mukherjee (2014). The collaborative effort between Earth scientists and microbiologists has produced success with remediation approaches using emulsified oils to promote enhanced reductive dechlorination or anaerobic degradation of chlorinated VOCs. Although many of the remedial efforts are in process and remediation goals have not been met, several sites have had significant reductions to VOC concentrations, and the progress is encouraging.

Challenges That Lie Ahead

Emerging contaminants and vapor intrusion may prove to be the next challenge for hydrogeologists working with contaminant assessment. Compounds like 1,4-dioxane and per- and polyfluoroalkyl substances (PFAS) are being considered by USEPA for inclusion to the list of regulated hazardous materials (USEPA 2018b). Toxicological assessments have identified 1,4-dioxane and PFAS at low concentration pose risk to human health and the environment, which raises public concern, thereby requiring regulatory evaluation.

1,4-dioxane commonly occurs with other chlorinated solvents because it was used as a stabilizer for those compounds. Because it is highly soluble, does not readily exchange from water to vapor, and does not readily sorb to carbon, it requires modifications to existing chlorinated solvent treatment systems and a reassessment of CSMs to project its advective migration pathways (USEPA 2018c).

PFAS present new challenges for those working in the contaminated groundwater industry because of the pervasiveness of items that could affect sample integrity. Teflon-containing equipment, waterproofed field gear, sun blocks, insect repellents—items that are typically used by samplers—are all potential sources of cross contamination (NGWA 2018). Considering that EPA has established a health advisory for PFAS compounds in drinking water at 70 parts per trillion (1×10^{-12}), it is likely that if standards are established, they will be at similar concentrations. Again, from a contaminant hydrogeology perspective, developing and understanding CSMs that accurately define fate, transport, and risk will be imperative. With PFAS, validating sample integrity will also be part of the overall assessment.

In the early years, hazardous waste impact assessments typically paid little attention to subsurface volatilization and potential risk to indoor air. However, vapor intrusion drew new attention in 2011 when USEPA finalized its reassessment of exposure to trichloroethylene (TCE) vapor and recognized the short-term exposure risks (i.e., 24-hour), particularly where there is the potential of exposure for women of childbearing age.

By 2016, states were reevaluating vapor intrusion policies, specifically where TCE was present in either soil or groundwater near residential sites. Many states reassessed the risk potential, and in some (e.g., Michigan; Gerstein 2018), residents were evacuated because of potential exposures. At sites where known TCE contamination exists, the USEPA Vapor Intrusion Screening Level calculator (VISL) was used to evaluate if the potential for exposure from groundwater to indoor air existed, and if the calculations showed the potential for vapor intrusion, assessments of sub-slab vapors and indoor air were accelerated. Now hydrogeologists place high priority on sites where TCE in groundwater is a potential pathway to indoor air.

Conclusions

Since the inception of the hazardous waste regulations, assessing contaminant hydrogeology has become a challenging, sometimes difficult, but often rewarding endeavor. Evaluating contaminant fate, transport, and risks are the focus of these professionals' efforts, and the emphasis on defining conceptual site models has expanded through the past decades along with advances in technology and capabilities.

The work completed by Bair and Svitana (2018) which looks at the Woburn Toxic Trial (the basis of the book and movie *A Civil Action*) from the lens of a teaching tool, also demonstrates the advances in contaminant hydrogeology since the 1980s. The website's visualizations page illustrates how the trial may have had a different outcome if the current state of the art in subsurface assessment and computer modeling were available in the 1980s (see the web link listed in Bair and Svitana [2018]).

As work on hazardous waste sites continues and technology advances, it is likely that our abilities to assess and validate risk will improve and continue to be an effective tool to demonstrate protection of human health and the environment.

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Environmental Isotope Tracers

By Ian D. Clark and Josué J. Jautzy

The hydrological cycle pumps over 1000 cubic kilometers of freshwater each day to the continents, of which more than 90% is transpired back to the troposphere by vegetation during photosynthesis. The balance runs off through the near surface or infiltrates into soils and aquifers. Tracing, groundwater recharge, subsurface pathways, and mixing can be resolved with stable isotopes and are the key objectives in the use of environmental isotopes.

The application of the natural isotopes ^{18}O and D now routinely accompanies hydrogeological studies. Moreover, an understanding of isotope fractionation and partitioning by Rayleigh distillation during rainout, evaporation, and recharge in the water cycle transfers directly to isotopes in a range of natural and anthropogenic compounds to resolve questions of contaminant sources, biodegradation, and transformation, where new analytical methods are providing important new tools. This chapter presents the fundamentals of isotope systematics in the water cycle and new approaches in contaminant hydrogeology.

Environmental isotopes

About 0.2% of oxygen nuclides carry an additional two neutrons, making ^{18}O a rare isotope of ^{16}O. About 0.015% of all hydrogen carries an additional neutron, making stable deuterium, ^{2}H or D, and together are conservative tracers for groundwater. Most elements have stable isotopes of varying abundance. Those of the biologically-active elements, H, C, N, O, and S, are very useful tracers of solutes in groundwater, providing insights on their origin as well as the geochemical reactions they have experienced in the subsurface. The stable environmental isotopes are those which lend themselves to easily analyzed and effective tracers of groundwater and its solutes. Others, while less routinely analyzed, are important tracers in more complex systems such as aquitards, deep crustal brines, and weathering (Table 1).

Stable isotope concentrations are measured as a ratio of the rare to the abundant isotope and rather than report them as absolute concentrations (e.g.,

Table 1. Stable environmental isotope tracers.

Isotope	*Ratio*	*natural abundance* %	*ppm*	*Reference (Coplen et al. 2006) (abundance ratio)*	*Common sample types*
D or ^{2}H	D/H	0.015	150	VSMOW ($1.5575 \cdot 10^{-4}$)	H_2O, CH_4, clays
^{13}C	$^{13}C/^{12}C$	1.11	11,100	VPDB ($1.1237 \cdot 10^{-2}$)	DIC, CO_2, $CaCO_3$, CH_4, DOC, SOC
^{15}N	$^{15}N/^{14}N$	0.366	3,660	AIR N_2 ($3.677 \cdot 10^{-3}$)	NO_3^-, NH_4^+, N_2, N_2O, SOC
^{18}O	$^{18}O/^{16}O$	0.204	2,040	VSMOW ($2.0052 \cdot 10^{-3}$)	H_2O, $CaCO_3$, O_2, NO_3^-, SO_4^{2-}
^{34}S	$^{34}S/^{32}S$	4.21	42,100	CDT ($4.5005 \cdot 10^{-2}$)	SO_4^{2-}, H_2S, gypsum, sulfide minerals
^{3}He	$^{3}He/^{4}He$	0.000138	1.38	AIR (0.00000138)	groundwater, minerals, gases
^{7}Li	$^{7}Li/^{6}Li$	92.4	924,100	LSVEC (12.17285)	water, brines, minerals
^{11}B	$^{11}B/^{10}B$	80.1	801,000	NBS 951 (4.044)	water, brines, carbonates
^{37}Cl	$^{37}Cl/^{35}Cl$	24.23	242,300	SMOC (0.324)	water, brines, TCE
^{87}Sr	$^{87}Sr/^{86}Sr$	7.0 and 9.8		Direct measurement	Sr^{2+} in water, brines, minerals

^{18}O in VSMOW is 0.002005 or 2005 ppm molar), they are expressed as the normalized difference (or delta, δ) between the sample and a known reference such as VSMOW (Vienna Standard Mean Ocean Water), express the measurement in parts per thousand or permil (‰) units:

$$\delta^{18}O_{sample} = \left(\frac{(^{18}O/^{16}O)_{sample}}{(^{18}O/^{16}O)_{VSMOW}} - 1 \right) \cdot 1000\,‰\ \ \text{VSMOW}$$

A δ-value that is positive, say $\delta^{18}O = +10‰$, signifies that the sample has 10 per mil or 1% more ^{18}O than the reference, and so is *enriched* in ^{18}O. Similarly, a sample with $\delta^{18}O = -10‰$ has 10‰ or 1% less ^{18}O than VSMOW, and so is *depleted* in ^{18}O.

Two important radioactive environmental isotopes bear discussion.

Tritium, T or ^3H, is a radioisotope of hydrogen that is produced naturally by cosmic radiation in the stratosphere and joins the active hydrological cycle as HTO. With its short half-life of only 12.32 years, it rapidly decays from natural levels of some 2 to 20 TU (tritium units, TU = 10^{-18} T atoms per H) in meteoric waters, and so is a useful tracer of modern recharge to groundwater.

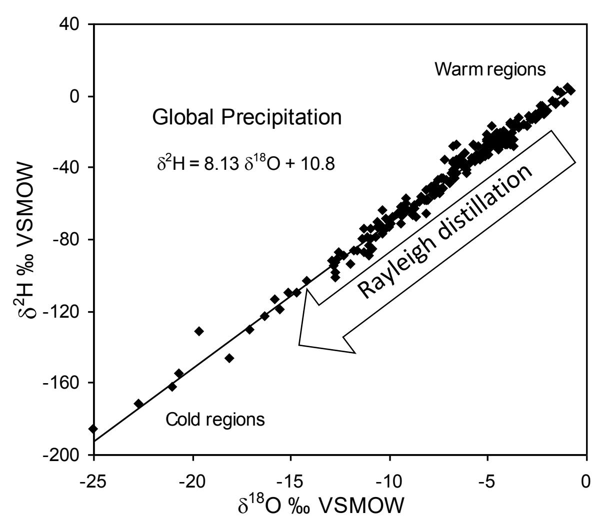
Radiocarbon, ^{14}C, is also atmospherically produced and is part of the active biosphere with a modern abundance ratio of 10^{-12} radiocarbon atoms per atom of stable carbon. Its long half-life of 5730 years allows dating of Holocene and late Pleistocene groundwaters using dissolved inorganic carbon ($DI^{14}C$) or dissolved organic carbon ($DO^{14}C$). It is increasingly used now as a tracer in organic contaminant studies, for example as a tracer of in-situ bioremediation by apportioning contributions of biodegraded hydrocarbon (^{14}C-free CO_2) in soil CO_2 emissions from natural soil respiration $^{14}CO_2$, or tracing fugitive gases from energy projects, and contaminant plumes.

Isotope partitioning in global freshwaters

While isotopes of a given element follow the same physico-chemical reactions, they do so at slightly different rates and different bond energies, such that there is a significant partitioning into the reactant or product reservoirs. The principal hydrological processes that affect the distribution of isotopes through the hydrological cycle are (i) evaporation and formation of atmospheric vapor over oceans but also from lakes and other surface water bodies, and (ii) condensation and rainout with decreasing temperature. Further, re-evaporation from soils and surface waters enriches the residual water in both isotopes, leaving a diagnostic signature on the groundwater that is preserved during recharge and subsurface flow. Finally, stable isotopes are conservative tracers, only rarely altered by isotope exchange with minerals and gases under extreme subsurface conditions such as in geothermal settings or in reactive materials with very low water to rock ratios and long residence times. In 1961, Harmon Craig published the earliest measurements of $\delta^{18}O$ and δD for freshwaters from around the globe. His two principal observations provide the foundation of isotope hydrology:

1. The strong linear correlation between $\delta^{18}O$ and δD in meteoric waters, with a slope of 8 and deuterium intercept of 10‰ (Figure 1)
2. Both ^{18}O and D in meteoric waters are enriched in warm regions and depleted in cold regions.

Figure 1. The $\delta^{18}O$–δD correlation for global precipitation plotted from data on the International Atomic Energy Agency GNIP database *(http://isohis.iaea.org).*



The observation of a strong correlation between $\delta^{18}O$ and δD provides the characteristic meteoric water lines for given regions that are used to determine the recharge input. The second observation that ^{18}O and D are partitioned into warmer regions reflects a general trend generated by a Rayleigh distillation of isotopes during rainout. These two observations are the basis of isotope hydrology. Further, this distillation of isotopes between product and reactant reservoirs is observed during biodegradation and many other environmental reactions.

The δ^{18}O–temperature correlation in precipitation

As precipitation is generated through cooling of the vapor mass, isotope fractionation between vapor and water or vapor and ice preferentially partitions ^{18}O and D into the rain or snow, such that successive rains become progressively depleted in ^{18}O and D along the vapor mass trajectory toward colder environments. This provides us with a series of temporal and spatial mechanisms to partition isotopes in water that is the basis of their use as tracers in hydrogeology. These are the global or latitudinal effect, the continental or distance from coasts effect, the elevation effect, the seasonal effect, and the paleoclimate effect.

The partitioning of isotopes between cold and warm regions is best observed on a global map of δ^{18}O in precipitation, shown in Figure 2, using mean annual precipitation data collected within the IAEA–World Meteorological Organization survey of precipitation. On global and regional scales, the temperature–δ^{18}O relationship is clear, with partitioning of ^{18}O into warmer, low-latitude precipitation and depletion in ^{18}O with increasing latitude.

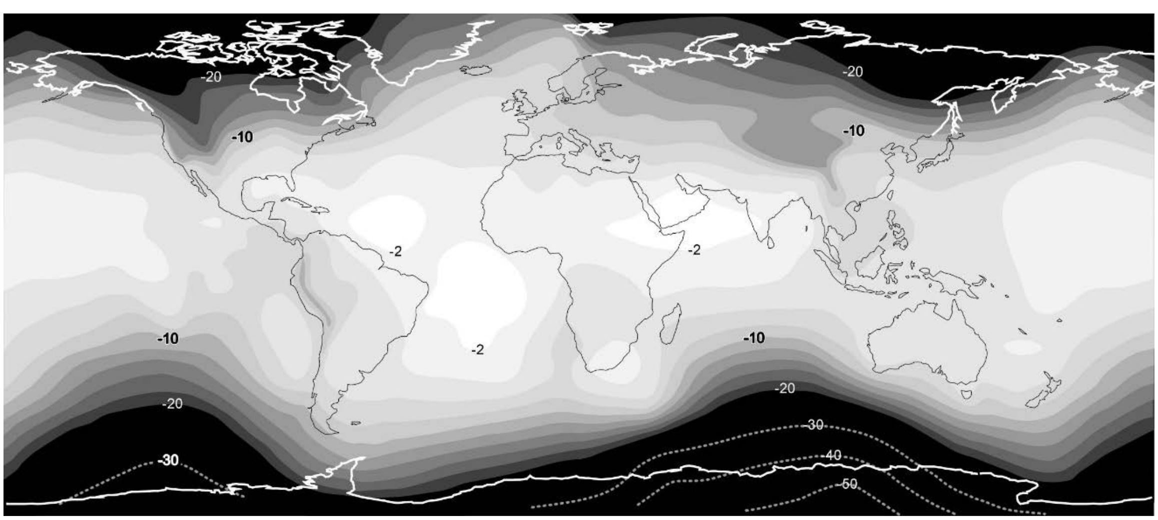
The very strong correlation between temperature and isotope content in precipitation observed on a global scale is manifested at the regional to local scale, where temperature gradients also occur. Even at the watershed scale, the decrease in temperature with elevation drives rainout and distillation of isotopes. For long-term data, distinctive isotope signatures emerge that are retained by groundwater recharged at different elevations. The lapse rate imparts a decrease in δ^{18}O of 0.15‰ to 0.5‰ per 100-m rise in altitude, while δD decreases by between 1 to 4‰. These differences can be observed not only in mountainous regions but also in catchments with only a few hundred meters of topography.

While seasonal signals in recharge are damped in the unsaturated flow, macroporosity can provide effective recharge to the water table, such that recharge from single storm events can be observed in phreatic aquifers through careful time-series sampling. Similarly, climatic shifts can be recorded, for example, by groundwaters recharged during glacial periods in northern regions (e.g., Grasby and Chen 2005) or during past pluvial times in currently arid regions (Clark et al. 1987).

Global and local meteoric water lines

The second observation by Craig (1961) is the strong correlation between ^{18}O and D in global freshwaters. The regression line for these data gives the "global meteoric water line" (GMWL), defined by the equation $\delta D = 8\,\delta^{18}O + 10$ ‰. The slope of 8 for this regression reflects the greater fractionation for D, which is about 8 times greater than that for ^{18}O. The IAEA maintains a global network of isotopes in

Figure 2. Global map of δ^{18}O for precipitation (from Clark 2015) based on data collected from International Atomic Energy Agency (IAEA) stations over the past 30 years. These long-term data are available at the IAEA website http://isohis.iaea.org/.



precipitation (www-naweb.iaea.org/napc/ih/IHS _ resources_gnip.html) with stations from all around the globe. These data include mean monthly values for $\delta^{18}O$ and δD together with precipitation amounts, temperature, and other meteorological data and are available for download. This provides local or regional precipitation data to produce local meteoric water lines (LMWL, Figure 3) for local groundwater studies. In any study of isotope hydrology, it is useful to use precipitation data that best represent the study area. In the absence of regional or local precipitation data, the GMWL is often substituted.

The $\delta^{18}O$–δD signal of precipitation can be modified in the recharge environment by evaporation during overland flow, for recharge from rivers and lakes, and even from unsaturated soils in arid regions. The loss of water by transpiration, in contrast, does not fractionate ^{18}O and D. The isotope effect of evaporation under conditions of lower humidity, non-equilibrium, or "kinetic" effects impart an additional fractionation with the result that evaporated waters plot characteristically below the meteoric water line on a $\delta^{18}O$–δD diagram, along an evaporation trend below the LMWL (Figure 3).

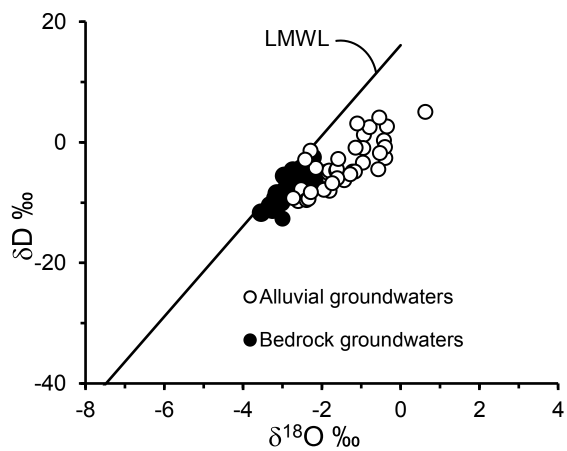
If evaporation is minor, then little to no effect will be observed in the residual water. If the water loss is more than a few percent, the result is a positive shift in the $\delta^{18}O$ and δD composition of the residual water away from a position on the local meteoric water line. This kinetic evaporation typically occurs during overland flow, in lakes and reservoirs, and from bare soils and sand deserts during infiltration.

Isotopes as contaminant tracers

With increasing anthropogenic pressure on groundwater systems, a need to look at the interactions and the persistence of contaminants in the subsurface has emerged. Organic components are of particular concern—including hydrocarbons, solvents, plasticizers, volatile organic compounds, pesticides, and low molecular weight polycyclic aromatic hydrocarbons. Isotope tracers can provide information on physico-chemical and biogeochemical processes in the subsurface and new techniques now focus on compound-specific analysis.

The pioneering work of Barrie et al. (1984) first measured the natural C-isotopic abundances at the molecular level. The challenge is that organic contaminants are often hidden by the larger amount of dissolved organic matter (DOM). A gas or liquid chromatograph interfaced with an isotope ratio mass spectrometer allows compound-specific isotope analysis (CSIA) on C-isotopes as well as N, H, O (Merritt and Hayes 1994; Burgoyne and Hayes 1998), and more recently on Cl, S, Br, and Hg isotopes using slightly different analytical strategies.

Figure 3. Evaporation effect observed in alluvial and bedrock groundwaters in an arid climate.



In the context of groundwater contaminant, CSIA has been extensively used in order to characterize degradation of the isotope enrichment factor (ε) in the laboratory. This parameter has been used to quantify degradation in natural systems using a Rayleigh model. While this technique provides good estimates of the evolution of the degradation of a contaminant plume in a groundwater system, limitations arise from inherent uncertainties due to sampling and analytical errors, variability of the source isotope ratios, and the accurate translation of the fractionation factor from the laboratory to the field.

In natural systems, CSIA on chlorinated solvents has helped differentiating between loss by migration/dilution or by degradation (Hunkeler et al. 2005; Sherwood Lollar et al. 1999). As different elements may behave differently relative to physico-chemical processes, CSIA has been subsequently extended to measurements of more than one isotopic system on the same molecule in groundwater systems.

For example, isotopes of C and Cl have been measured on chlorinated solvent in a contaminated aquifer to assess the extent of dechlorination (Wiegert et al. 2012). In laboratory settings, this two-dimensional CSIA applied on C and H isotope systems showed its useful potential in discriminating between benzene transformation pathways (Fischer et al. 2008). Recently this technique has been applied to the catchment scale to decipher pesticides export losses from degradation in a near-surface hydrological context (Alvarez-Zaldívar et al. 2018).

In addition to these legacy contaminants, a relatively new type of micro-contaminant is now also studied. Micro-contaminants are ubiquitous at trace level (ng.L^{-1} to µg.L^{-1}) in groundwater systems and are derived from pesticides, pharmaceutical compounds, and consumer care products. Due to their relatively high polarity, alternative techniques for compound separation such as liquid chromatography or derivatization are used. Moreover, specific analytical challenges are associated with their isotopic analyses such as the quantity of sample required to reach sufficient mass for CSIA. This sampling design also increases the potential of matrix interferences in the organic extract and therefore requires particular purification strategies.

While CSIA provides a means of targeting natural changes in isotopic composition on specific molecules, it has been shown early on that the *intra*molecular isotopic abundance distribution within molecules was heterogeneous (Abelson and Hoering 1961). This heterogeneity is inherited from specific pathways of formations or through degradation on specific active sites. This higher resolution of isotopic information has been recognized as an untapped resource for the study of groundwater contamination (Elsner et al. 2005).

The position-specific isotopic signature is often diluted in the large number of isotopically invariant sites and requires sensitive preparative chemistry to be able to analyze molecular fragments separately. With the recent advances in high- and ultra-high-resolution mass spectrometric techniques, it is likely that diverse areas of research will benefit from this potential high throughput position-specific isotope analysis including hydrogeochemistry and its use in tracking the sources and fate of contaminants within groundwater systems.

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Groundwater: State of the Science and Practice

Fractured Rock Environments

By Paul A Hsieh

Fractured rocks are typically highly heterogeneous. For example, hydraulic conductivity (K) can vary by orders of magnitude over distances as short as a few meters. Heterogeneity poses a challenge to investigating groundwater flow and contaminant transport. Hydraulic properties (K and S_s, specific storage) measured at one location might not be representative of other locations, even if nearby. Analysis techniques based on the assumption of homogeneity (e.g., analytical solutions for aquifer test) might be inadequate. A high degree of heterogeneity could manifest as a scale effect: several studies reported that, on the aggregate, K's measured at smaller scales (e.g., by single-hole tests) tend to be smaller than K's measured at larger scales (e.g., by cross-hole tests or by model calibration).

In a situation of groundwater pollution, contaminants could rapidly move along fast pathways to spread far from the contaminant source. Conversely, contaminants might be trapped in near-stagnant regions, thus making cleanup difficult. Knowledge of heterogeneity is therefore critical for effectively managing and protecting groundwater resources in fractured rock environments. Inadequate characterization of heterogeneity could result in unreliable model forecasts.

The strong heterogeneity in fractured rocks, however, does not signify a wholesale change is required in study approach. Given that the basic principles of fluid flow, solute transport, and biogeochemical processes apply equally to both porous and fractured media, established methods of groundwater study still form the backbone for fractured rock investigations.

For example, understanding the geological framework, geophysical exploration, monitoring hydraulic head, measuring inflows (e.g., recharge) and outflows (e.g., baseflow to streams that drain the aquifer), well testing, and geochemical sampling all remain essential activities. However, adequate characterization of heterogeneity of fractured rocks might require a higher level of data collection as well as the use of additional specialized techniques.

Data interpretation would likely require numerical modeling and parameter estimation (inverse modeling). Utilizing a broad variety of tools could lead to a more robust understanding of a fractured rock environment. Taking an interdisciplinary approach is the first recommendation in the report titled "Characterization, Modeling, Monitoring, and Remediation of Fractured Rock" by the National Academies of Sciences, Engineering, and Medicine (2015).

An overview of geophysical technologies appropriate for fractured rock investigation is presented by Day-Lewis et al. (2017). Running a suite of borehole geophysical logs using modern instruments yields high-resolution information on rock lithology, fracture locations and orientations, and rock properties such as acoustic wave velocity that might be related to fracturing. By measuring the distribution of groundwater inflow along a pumped borehole, a flowmeter survey can provide transmissivity estimates for individual fractures or groups of closely spaced fractures.

To probe beyond the near vicinity of a wellbore, cross-hole geophysical methods have been developed to construct 2D or 3D images of rock properties between boreholes. Repeated application can produce time-lapse images to monitor, for example, the change in electrical conductivity as an ionic tracer is injected into the rock. Cross-hole imaging methods are only now transitioning from research to application and require sophisticated processing software. As noted by Day-Lewis et al. (2017), geophysical methods commonly yield results that are indirectly related to quantities of hydrologic interest (e.g., K or solute concentration). Joint inversion of geophysical and groundwater flow/transport models might lead to a more definitive interpretation of both geophysical and hydrologic data.

Determining the age of groundwater by measuring the concentrations of environmental tracers (e.g., CFC-113, SF^6, ^3H, and ^3He) provides an approach for predicting contaminant travel time in the subsurface and for calibrating groundwater models. Such applications, however, are challenging to apply for fractured rocks due to highly convoluted flowpaths arising from the juxtaposition of high-K and low-K regions. The spatial distribution of groundwater age might be difficult to interpret. For example, younger water might underlie older water. The exchange of solute between regions of flowing and stagnant water adds another layer of complexity to the transport process.

In principle, concentration of environmental tracers can be simulated with a full-featured solute transport model incorporating the complex distribution of groundwater velocities. However, such an effort would require intensive computational resources. Consequently, simpler approaches have been adopted, such as piston flow (no dispersion), binary-dilution (mixing of old and young waters), advection along individual flowpaths, and transport through dual-domain (consisting of mobile and immobile regions).

Sanford et al. (2017) compared these four methods for interpreting environmental tracers sampled from springs discharging from fractured carbonate rocks in Virginia and West Virginia. They concluded the dual-domain method provided the best match to their tracer data, and suggested that the combined use of multiple environmental tracers with the dual-domain approach could be applicable in a wide variety of fractured rock settings.

Hydraulic testing (pumping tests, packer tests, etc.) remains an essential method for determining hydraulic properties in the field. Packers can be used to isolate individual intervals in wells, thus enabling testing in a 3D configuration. Single-well tests, although simpler to carry out, characterize only the near vicinity of the borehole. Cross-hole tests characterize a larger volume of rock mass, but require more effort and equipment. In the presence of heterogeneity (e.g., presence of high-K zones), numerical modeling is often necessary for data analysis.

Case histories (mostly at crystalline rocks sites) suggest that the spatial pattern of K and S_s in the model need not be highly complex. For example, Martinez-Landa et al. (2016) analyzed a short-term (<1 day) cross-hole test in granitic rock using a numerical model containing eight structural features (e.g., faults, dykes, and major fractures). However, they emphasized the importance of identifying and characterizing these features using structural geology and geophysical methods. The calibrated model was then able to successfully predict drawdowns during a long-term (4-month) large-scale pumping test.

In recent years, groundwater researchers are also developing a high-resolution testing method known as hydraulic tomography (Illman 2014). This method involves a multitude of cross-hole tests using many combinations of pumping/injection and observation intervals in wells. Although the field effort and computational requirement are demanding, the high-resolution images of K and S_s produced by hydraulic tomography would be warranted when knowledge of these properties are of critical importance.

Approaches to modeling flow in fractured rocks have traditionally been divided into the continuum approach (also known as the equivalent porous medium approach) and the discrete fracture network (DFN) approach.

A continuum model is similar to a conventional groundwater model in that the aquifer is characterized by the spatial distribution of K and S_s, which might be (1) composed of zones with K and S_s being uniform in a zone but different from zone to zone, (2) spatially varying according to a functional form (e.g., K decreasing with depth), or (3) generated by stochastic methods. By contrast, a DFN model simulates flow through individual fractures in a fracture network, usually constructed by embedding known major features within a stochastically generated network of fractures.

Both approaches are now firmly established in practice. Hybrid approaches have also been used whereby a DFN is converted into a K field for continuum modeling. Less commonly used, the channel network model is yet another modeling approach. This model is based on the concept that, in a naturally rough fracture, flow is not uniformly distributed over the fracture plane, but is focused along channels composed of more transmissive regions.

Comparisons of modeling approaches (e.g., Ko et al. 2015) show that no one approach is inherently superior to another in the ability to simulate observed data such as hydraulic head. Modeling studies of fractured rock sites suggest that, regardless of model choice, adequate representation of heterogeneity is a key to building a useful model. For a given site, the major structural features such as faults or high-K zones would have to be identified, characterized, and explicitly incorporated into the model. Conversely, the background network of minor fractures would be represented as a continuum or as a stochastically generated fracture network.

Remediating contaminated fractured rock sites remains a difficult undertaking. Studies in recent years show matrix diffusion can be an important mechanism of contaminant storage in the subsurface, especially for fractured sedimentary rocks with significant porosity in the matrix blocks. As a dissolved contaminant moves with the groundwater through transmissive fractures, a portion of the contaminant diffuses from fractures into the pore space in the matrix blocks. The contaminant might also sorbed onto rock grains in the matrix. Over a long period of time (e.g., tens of years), a large amount of contaminant can accumulate in the matrix. Even if the contaminant source at the surface is later removed, the contaminant stored in the matrix would slowly desorb and diffuse back into the flowing groundwater. The site could remain contaminated for tens to hundreds of years. Thus, an import technique to investigate such contaminated sites is sampling for contaminants in drill cores (Parker et al. 2018).

Predicting the fate and transport of contaminants requires detailed knowledge of contaminant distribution. Effective application of in situ remediation technology requires detailed knowledge of flowpaths so that treatment fluids or amendments can be delivered to locations where contaminants are stored.

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Geophysics (Surface/Aerial/Subsurface)

By John Jansen, Ph.D., PG, PGp, Doug Laymon, PG, and Finn Michelsen, PG

Geophysics is the science of using remote sensing to interpret the physical properties of the subsurface. Geophysical methods measure ambient or transmitted natural fields to estimate physical properties of the subsurface that can be used to infer other properties of interest. An example of this is using electromagnetic waves to measure the electrical conductivity of a volume of material and inferring the grain size or fluid properties based on the measurement.

Geophysical methods can be used to fill data gaps and collect data in a non-invasive manner when conducting groundwater studies. The remote sensing methods are usually not as reliable or unambiguous as direct observations such as drilling or digging. However, there are many situations where direct measurements are not possible or it is impractical to get enough coverage through direct investigation alone. In those cases, adding geophysical methods can help correlate units between borings, pinpoint formation boundaries and stratigraphic pinch-outs, detect anomalous bodies or features, find fractures or faults, measure mechanical properties, and provide a higher degree of confidence that the critical features at a given site have been identified and characterized. It is also possible to obtain information about the pore fluid of a formation such as salinity or the presence of non-aqueous phase liquids (NAPLs).

Figure 1. Electrical resistivity tomagraphy and seismic retraction tomography to map karst features.

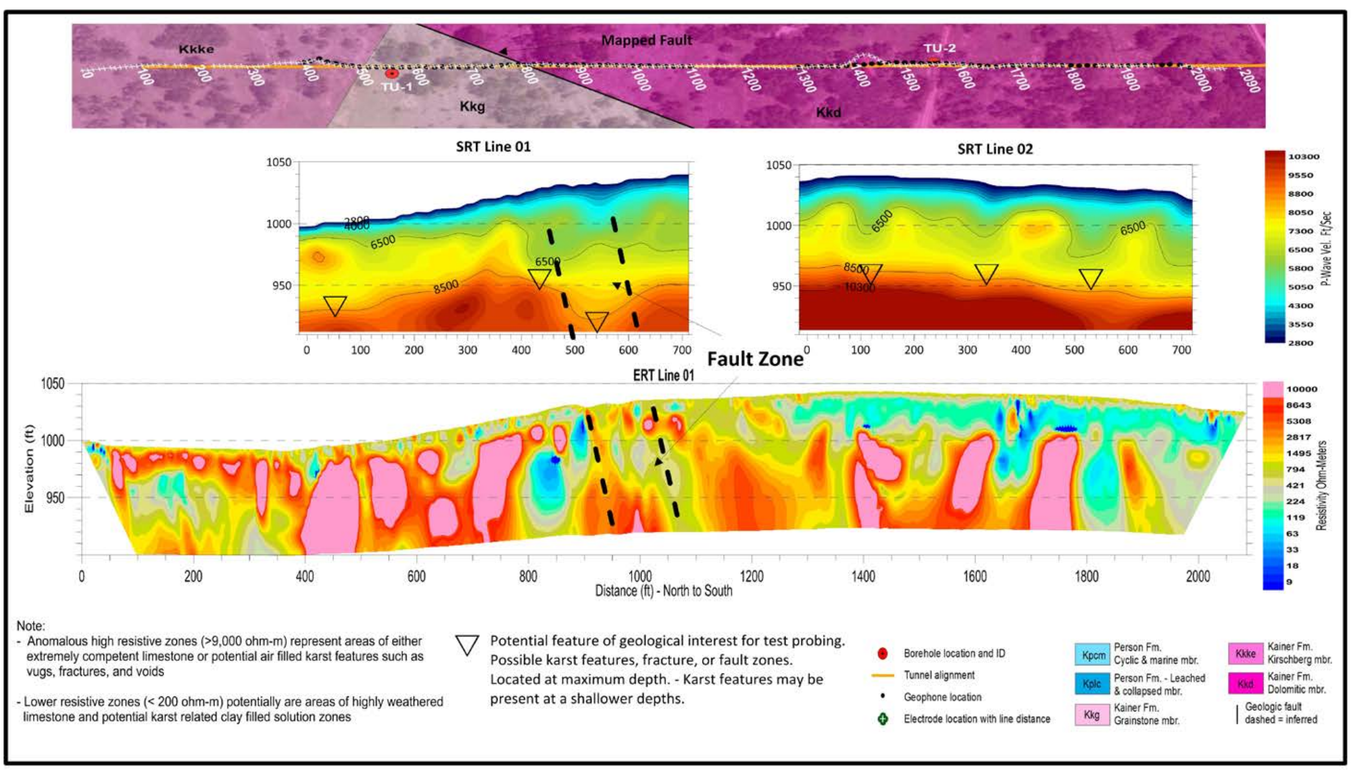
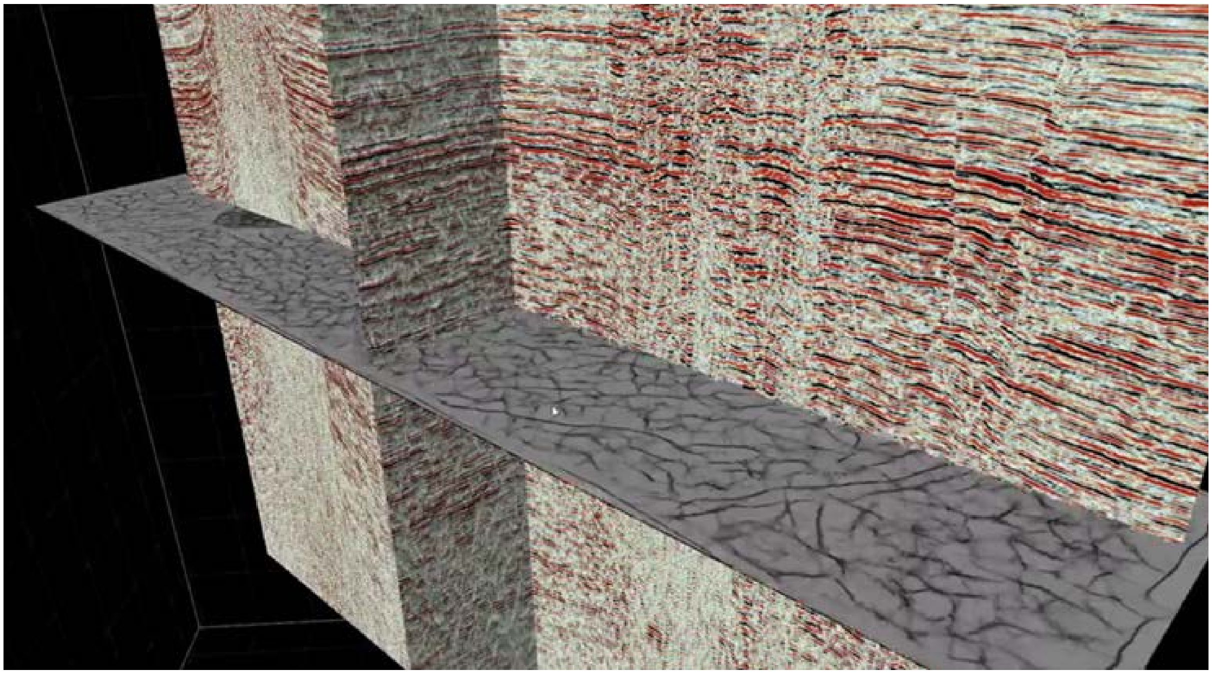


Figure 2. 3D seismic reflection data cube showing vertical perpendicular seismic lines and faults and fractures in a horizontal plane.



The resolution of the various geophysical methods varies from several tens of feet for deep studies using electromagnetic induction methods to fractions of an inch using ground penetrating radar with high frequency antennas. The resolution of the geophysical method used must be matched to the physical dimensions of the target body if the target is to be detected. The magnitude and size of the anomaly must be predicted with reasonable accuracy, typically by forward modeling, to determine the minimum grid size that can be used to have a reasonable probability of detection.

Site conditions also play an important role in determining the ultimate success of the survey. Surface and subsurface conditions can create a noisy environment that can easily obscure the signal from the target body. Sources of noise are different for each method. For instance, ground vibrations from traffic or machinery can be a major source of noise for seismic or gravity methods, but will not affect other methods such as electromagnetic induction or magnetics.

Physical conditions at a site also affect the utility of geophysical methods. Major obstructions such as buildings or developed land often limit the area where data can be gathered and create inherent limitations on the geophysical survey. Buildings, parked cars, dumpsters, overhead power lines, buried utilities, fences, buried rubble, and other cultural features must be considered when designing a geophysical survey because they will determine where you can make valid measurements. In addition, subsurface conditions such as soil type, uniformity, and other factors affect the propagation of energy and can limit the performance of most methods. Surface conditions such as paved surfaces, heavy vegetation, frozen ground, rough topography, and surface debris must also be considered when designing a survey.

Geophysical measurements can be made using survey equipment on the earth's surface (surface methods), deployed from aircraft (airborne methods), or using probes trolled up and down boreholes (borehole methods). Some methods can be deployed on water bodies, both from the surface or the bottom, and in both freshwater and seawater. Several methods are being modified to be deployed on small unmanned drones, both airborne and waterborne. The use of drones promises to increase the data density that can be obtained and allow surveys to be conducted in areas that are otherwise inaccessible.

Methods can also be classified as active or passive measurements. Methods that measure variations in natural fields, such as gravimetry or magnetometry, are called passive methods because they use natural fields propagating through the earth. Other methods measure the response of material to some form of transmitted energy and are called active methods because they actively transmit the energy used to probe the subsurface. Active methods include most seismic methods, most electrical and electromagnetic methods, and most downhole porosity tools.

No matter how the sensors are deployed, they are generally making equivalent measurements, though the differences in their method of deployment and local environment around the sensors cause significant differences in the sensitivity, depth of investigation, and susceptibility to interference of the measurements.

Surface measurements can be made as one dimensional (1D) soundings that assume laterally continuous layers, as two dimensional (2D) profiles along survey lines, or as three dimensional (3D) volumes using a grid of surface measurements or between surface points and boreholes. Borehole measurements can be made within a single borehole or as tomographic measurements between multiple boreholes.

Measurements can also be made in the fourth dimension (4D) of time where fixed sensor arrays are used to collect data at various intervals in time to monitor temporal changes in the subsurface over a fixed period of time. Examples of changes that can be monitored in 4D include the presence and movement of groundwater and changes in the chemistry of groundwater.

Most geologic materials have characteristic physical properties that can be used to identify them with geophysical data. Clay has lower electrical resistivity and higher seismic velocity than sand. Bedrock generally has a higher seismic velocity and density than unconsolidated materials. Hard rock like carbonates and igneous or metamorphic bedrock has higher seismic velocity and density than softer rocks like shale or sandstone. Saturated materials generally have lower resistivity values than unsaturated material. Materials saturated with fresh groundwater generally have higher resistivity than the same material saturated with brackish or saline water. Many of these methods are commonly used to find man-made objects like utilities, storage tanks, or unexploded ordnance. These types of surveys are designed for shallow penetration and high resolution with very dense sampling density and may use specialized processing algorithms.

The major energy fields used by geophysical methods include magnetic fields (magnetometry); electrical fields (electrical resistivity and spontaneous potential); electromagnetic fields (electromagnetic induction and ground penetrating radar); propagation of seismic waves (seismic reflection, seismic refraction, passive acoustic emission monitoring, and spectral analysis of surface waves); the gravitational field (gravimetry); gamma ray radiation (gamma ray spectroscopy); and heat transfer (geothermal). Other physical properties can be measured by bombarding the material with gamma rays (electron density) or high energy neutrons (hydrogen content). Less commonly, properties such as thermal conductivity, electrical chargeability, or magnetic resonance are also measured.

Data visualization and reporting are key components of any geophysical survey. Appropriate communication and clear reporting of the survey results are important to properly integrate the geophysical results into the overall solution for the project. Data presentation and visualization is also an important tool in communicating the results of geophysical data sets. Geophysical data can be integrated with other site data using GIS (geographic information system) and high-resolution color contour packages. Data are often presented in 2D plan view contour maps. If the application calls for it, 3D data presentation is utilized for visualization of complex geophysical data sets. Additionally, key targets and anomalies are commonly called out on interpretive maps.

Geophysical measurements are indirect. The more you know about the site's physical characteristics and hydrogeology beforehand, the better you can choose a method that focuses on a property that will give useful information. Geophysical measurements have an inherent level of uncertainty that can be minimized, but not eliminated. Using multiple geophysical methods that make independent measurements or measure different physical properties will significantly increase the sensitivity and reliability of a geophysical survey.

There are a large number of geophysical methods that have been developed over the past 100 years. Descriptions of the variety and application of these methods fill many books and can take a lifetime to master just a small subset of the field. With that in mind, Table 1 provides a listing of the major methods with some cursory information on the measurement and applications. This table is best used as a checklist to direct the reader toward more research of potential methods for a given objective.

Table 1. List of common geophysical methods and applications.

Table 1: List of Common Geophysical Methods and Applications			
Method	**What it Measures:**	**Mode of Application**	**Typical Uses**
ELECTRICAL METHODS			
Electrical Resistivity	Electrical Conductivity	Surface & Marine	Stratigraphy, Saltwater Intrusion, Fracture Zones
Induced Polarization (IP)	Electrical Chargeability	Surface	Sulfide Mineralization, Clay Content
Spontaneous Potential (SP)	Electrokinetic Potential	Surface	Fluid Flow
Mise a la Masse	Electrical Conductivity	Surface	Conductive Bodies
SEISMIC METHODS			
Seismic Refraction	Seismic Velocity	Surface	Depth to Bedrock or Confining Units
Seismic Reflection	Acoustic Impedance	Surface & Marine	Stratigraphy, Structure, Faulting
Multi-Channel Analysis of Surface Waves (MASW)	Shear Wave Velocity	Surface	Depth to Bedrock, Voids, Incometent Zones
Full Wave-Form Tomography	Seismic Wave Propagation	Surface	Stratigraphy, Structure, Karst
Horizontal to Vertical Spectral Ratio (HVSR) Method	Shear Wave Velocity	Surface	Depth to Bedrock
ELECTROMAGNETIC METHODS (EM)			
Frequency Domain Electromagnetic Induction (FDEM)	Electrical Conductivity	Surface, Marine & Airborne	Stratigraphy, Saltwater Intrusion, Fracture Zones
Time Domain Electromagnetic Induction (TEM)	Electrical Conductivity	Surface, Marine & Airborne	Stratigraphy, Saltwater Intrusion, Fracture Zones
Ground Penetrating Radar (GPR)	Dielectric Constant	Surface & Marine	Stratigraphy, Buried Targets
Controlled Source Audio Frequency Magnetotellurics (CSAMT)	Electrical Conductivity	Surface	Stratigraphy, Saltwater Intrusion, Fracture Zones
Very Low Frequency Induction (VLF)	Electrical Conductivity	Surface	Bedrock Fractures, Depth to Bedrock
Metal Detectors	Electrical Conductivity	Surface & Marine	Buried Metal, Utilities
POTENTIAL FIELD METHODS			
Magnetometry	Magnetic Susceptibility	Surface, Marine & Airborne	Ferrous Bodies
Gravity Surveys	Density	Surface, Marine & Airborne	Depth to Bedrock, Voids, Structure
Geothermal Methods	Thermal Conductivity	Surface	Fluid Flow
BOREHOLE METHODS			
ELECTRICAL LOGS			
Spontaneous Potential Log	Electrokinetic Potential	Fluid Filled Borehole	Sand vs Shale, Water Quality
Resistivity Logs	Electrical Conductivity	Fluid Filled Borehole	Stratigraphy, Water Quality
Resistance Logs	Electrical Resistance	Fluid Filled Borehole	Formation Contacts
Induction Logs	Electrical Conductivity	Fluid or Air Filled Borehole	Stratigraphy, Water Quality
Gamma Logs	Gamma Ray Emission	Fluid or Air Filled Borehole	Clay Content
POROSITY LOGS			
Nuclear Magnetic Resonance Log (NMR)	Hydrogen Ion Content	Fluid or Air Filled Borehole	Porosity and Permeability
BOREHOLD IMAGING LOGS			
Down Hole Televising Log	Borehole Image	Clear Fluid or air Filled Borehole	Borehole Condition, Stratigraphy
Acoustic Televiewer	High Frequency Sonic Scan	Fluid Filled Borehole	Borehole Condition, Fractures
Optical Televiewer	Optical Light Scan	Clear Fluid or Air Filled Borehole	Borehole Condition, Stratigraphy, Fractures
Caliper Log	Borehole Diameter	Any Borehole	Borehole Diameter, Fractures
Alignment Logs	Borehole Deviation	Any Borehole	Hole Alignment
FLOW METERS			
Temperature Logs	Fluid Temperature	Water Filled Borehole	Flow in Open Borehole
Borehole Fluid Conductivity Logs	Electrical Conductivity	Water Filled Borehole	Flow in Open Borehole
Spinner Logs	Fluid Flow	Water Filled Borehole	Flow in Open Borehole
Heat Pulse Flow Meters	Fluid Flow	Water Filled Borehole	Flow in Open Borehole
Electromagnetic Flow Meters	Fluid Flow	Water Filled Borehole	Flow in Open Borehole
Fluid Displacement Logs	Fluid Flow	Dionized Water Filled Borehole	Flow in Open Borehole
WATER QUALITY LOGS			
Geochemical Logs	Ionic concentration	Water Filled Borehole	Concentration of Specific Ions like Chloride or Nitrate
Downhole Samplers	Water Quality	Water Filled Borehole	Collecting Water Samples from Specific Depths in a Water Filled Borehole

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Groundwater and Unconventional Oil and Gas Deveopment

By Daniel J. Soeder

Introduction

The successful production of hydrocarbons from "unconventional" resources like shale has opened new energy reserves in North America and worldwide, although the hydraulic fracturing technology ("fracking") required for economical recovery poses both real and perceived risks to groundwater (Soeder and Kent 2018). Geophysical data show the tops of hydraulic fractures remain many kilometers below drinking water aquifers (Warpinski 2013), and field studies with tracers provided no evidence supporting the perceived risk of upward migration (Hammack et al. 2014). Shallow oil and gas resources close to drinking water aquifers are not fracked because of insufficient overburden stress (Soeder 2017). Actual risks from fracking include both water availability and water quality issues (U.S. Department of Energy 2015).

State of the Science

In the early days of shale development, operators often obtained frack water from municipal utilities or used local groundwater, directly impacting drinking water supplies. Most shale developers now use proper management with more sustainable sources like large rivers or non-potable groundwater.

The two primary water quality risks to groundwater from unconventional oil and gas are (1) stray gas and (2) contamination from fluid or chemical spills. Stray gas is produced through a variety of biogenic or geochemical processes within an aquifer, and can migrate from adjacent geologic units like coals or enter aquifers via wellbore leakage (Townsend-Small et al. 2016). Defining a stray gas source requires molecular and isotopic geochemical data (Baldassare et al. 2014). Methane is not considered a groundwater contaminant per se, in that the gas is not hazardous to drink, but it can accumulate in confined spaces to produce explosions.

Osborn et al. (2011) concluded stray gas concentrations in northeastern Pennsylvania groundwater increased in the vicinity of shale wells. Molofsky et al. (2013) determined methane was ubiquitous in groundwater in this area, and linked concentrations to topography. Siegel et al. (2015) analyzed a massive database of groundwater samples from northeastern Pennsylvania and concluded no statistically valid correlation exists between methane in groundwater and proximity to gas wells. Ingraffea et al. (2014) investigated Pennsylvania state incident reports and found a higher frequency of wellbore integrity failures associated with horizontal shale gas wells compared to vertical conventional wells. Methane in aquifers above gas shales has been found to be mostly unrelated to natural gas in the underlying shales, suggesting upward migration is uncommon (Rivard et al. 2016; McMahon et al. 2017).

The other major water-quality risk to groundwater from oil and gas development is a chemical spill (Soeder et al. 2014). Water-based drilling fluids stored in lined pits on drill pads often leak and allow the fluids to infiltrate into the ground (Figure 1). Oil-based drilling fluids have better performance and have become popular in shale plays, but are also considerably more expensive, giving operators incentives to use leak-proof tanks.

Figure 1. Seepage of water-based drilling fluid from a pit through shallow soil and into Indian Run, West Virginia.

Photo in 2009 by adjacent landowner Doug Mazer, used with permission.

Frack fluid consists of water with additives such as polyacrylamide to create friction-reducing "slickwater" and other chemicals to inhibit corrosion, mineral precipitation, and downhole bacteria growth. Thousands of gallons of chemicals are transported to shale well pads and stored on-site until used, posing a risk for spills or leaks. Many chemicals are proprietary, and new ones are constantly being added. Little is known about the fate and transport of these chemicals in groundwater (Kahrilas et al. 2016). Polyacrylamide, for example, degrades to acrylamide, a reproductive toxin and carcinogen (Exon 2006)

Along with hydrocarbons, fluids recovered from a shale well include some of the water and chemicals injected downhole, plus naturally-occurring formation brines, which often have concentrations of dissolved solids many times greater than seawater. Most of the risks associated with produced water are related to the transport and disposal processes. Disposal of most wastewater from both conventional and unconventional wells by deep underground injection can result in induced seismicity (Rubinstein and Mahani 2015). Vehicle accidents and pipeline breaches during wastewater transport (Cozzarreli et al. 2017) and spills or leakage from careless handling of waste liquids (Akob et al. 2016) also pose groundwater risks.

Practices

Pre-development baseline data on levels of methane, dissolved solids, and organics in aquifers are needed for understanding the impacts of hydraulic fracturing on groundwater quality. Investigators seeking to evaluate changes in groundwater chemistry by sampling active shale gas development sites (i.e., Barth-Naftilan and Saiers 2015) often lack baseline data collected before development (Figure 2) to provide unequivocal evidence linking certain contaminants to shale gas production.

Figure 2. Interns from the DOE National Energy Technology Laboratory collect pre-drilling baseline data from a spring in Moshannon State Forest, Pennsylvania, in 2014.

Photo by Dan Soeder.

Difficulties accessing field sites, samples, and industry data pose significant challenges to non-industry researchers studying the environmental risks of shale gas (Soeder 2015). The state of the art requires a properly characterized field site with dedicated monitoring wells equipped with multilevel samplers to understand how gas and liquids associated with shale wells migrate through shallow aquifers (Cherry et al. 2015). Such field-based research has been recommended by various committees and academic reviewers (e.g., Jackson et al. 2013) but has rarely been done. Efforts to develop standard sampling and analysis practices to compare results between different shale gas environmental studies are also needed, along with a better understanding of the natural attenuation paths for frack chemicals.

The investigations required to assess the risks for groundwater contamination from unconventional oil and gas are neither short nor simple. Linkages between well construction practices, frack designs, and groundwater quality are needed to understand how a particular procedure might release chemicals or stray gas. Researchers must gain access to industry data on well construction, accidents, and failures to truly understand risks.

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Groundwater Depletion

By Ken Rainwater, Ph.D., P.E., BCEE, D.WRE, CFM

Aquifer Storage Concepts

While aquifers may be laterally extensive due to their geologic origins, these subsurface water-bearing zones must be recognized as finite storage volumes for many of our most precious freshwater resources. Understanding groundwater depletion requires a complete water balance for the target aquifer. Total aquifer storage volume calculations are based on maps of variations in saturated thickness coupled with estimates of storage coefficients. Changes in storage can then be estimated with the areal changes in water levels, accurate representation of the saturated thickness, and good estimates of storage coefficients.

Outflows from the aquifer can include spring discharges, evapotranspiration, leakage to vertically adjacent aquifers, and withdrawals from pumping wells. Springs may be difficult to locate and observe quantitatively. Evapotranspiration can be estimated by plant location and type, root depth, and growing season. Leakage to other aquifers can be estimated based on head differences between multiple aquifer layers and the local vertical hydraulic conductivities. Pumped withdrawals can be metered and reported as part of groundwater management programs, but this approach is not universal.

Inflows include recharge downward through the unsaturated zone above the water table, leakage from a nearby layer with higher head, and artificial recharge. Natural recharge rates are typically inferred from estimates of infiltration past the local root zone. In addition, the time of vertical travel through the unsaturated zone is controlled by the vertical distance from the surface to the local water table and the hydraulic and soil characteristics, which may take decades for sedimentary systems or days for karst systems. Precipitation intensity and duration combine with the local topography, soil, and vegetation distributions to impact the temporal variations in recharge.

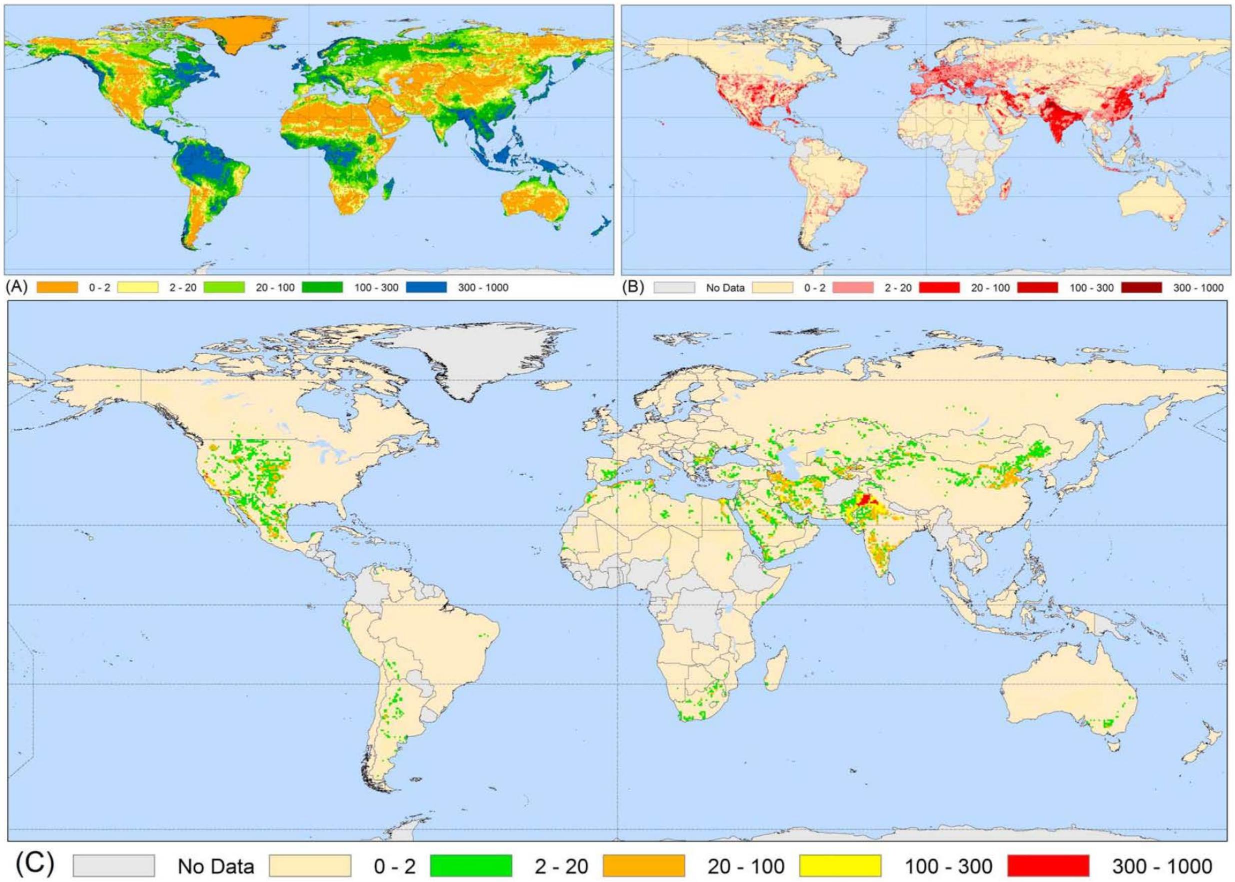
The chloride mass balance approach is popular for long-term average estimates in recharge rates, as the relatively fresh infiltrating rainfall can cause the salts in the root zone left behind by evapotranspiration to move downward. Numerical models are also used, either as a direct vertical unsaturated flow model like HYDRUS 1-D or as a calibration result for a regional groundwater flow model.

Using the water balance for the aquifer, depletion occurs when the sum of the inflows is less than the sum of the outflows. The most typical depletion situation is when many nearby pumping wells remove more water than can be replenished by the recharge across that area. Gleeson et al. (2012) proposed the groundwater footprint (GF) concept to indicate aquifer stresses when abstractions exceed the net input to the aquifer expressed as recharge minus environmental discharges to springs. The concept was applied with a large-scale global hydrologic model and grid-based estimates of withdrawals, and required no estimates of storage changes.

Observing Depletion

Aquifer depletion is typically documented by increasing depth to water readings over time at observation wells distributed across all or part of the aquifer's areal extent. Normal practice is to collect these data a few to several months after the end of the water well-based irrigation season, referred to by some as "static" water levels even though they might be changing year to year. Single measurements may be taken with e-lines or sonic sounders, while more continuous data can be collected with pressure transducers with dataloggers, and contour maps can be constructed with appropriate software packages.

Figure 1. (A) Simulated average groundwater recharge by PCR-GLOBWB, (B) total groundwater abstraction for the year 2000, and (C) groundwater depletion for the year 2000 (all in mm/yr) (Wada et al. 2010).



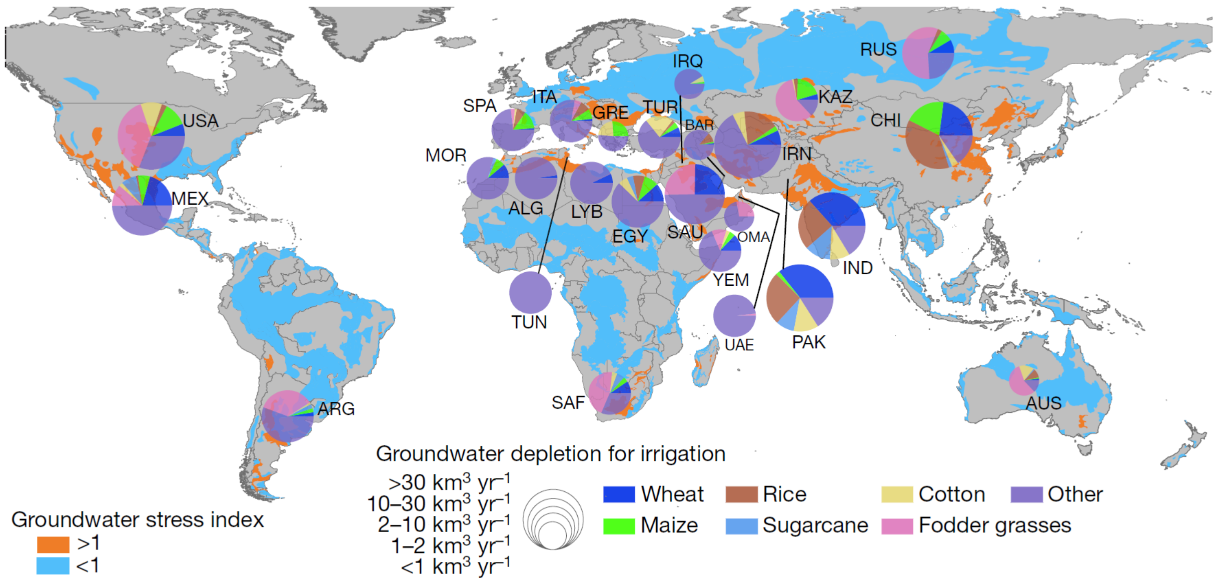
The data can also be used in regional and larger-scale groundwater models as done by Wada et al. (2010) to generate a global map of groundwater depletion, as shown in Figure 1 for the year 2000. More recently, researchers applied NASA's Gravity Recovery and Climate Experiment (GRACE) satellite system to monitor changes in the Earth's gravity field that identify time-variable anomalies in terrestrial water storage in snow, ice, surface water, and groundwater (Richey et al. 2015). This impressive application of remote sensing is useful for locations that do not have local water level monitoring programs, but must be combined with local ground-truth observations for locally precise evaluations.

Another potential indicator of local groundwater depletion is water quality. For many thick aquifers, the deeper water is more saline and dense due to the groundwater's age. The lower quality water may require treatment before municipal or agricultural uses. Finally, depletion of confined aquifers can lead to significant land surface subsidence, as has been seen in the Houston vicinity and California's Central Valley.

Many researchers and government agencies have documented depletion in major aquifers in the United States. For example, Scanlon et al. (2012) considered the High Plains (Ogallala) and Central Valley aquifers, comparing the spatial distribution of recharge rates estimated through the chloride mass balance approach with observed storage changes and regional groundwater flow model simulations. These efforts then provided estimates of remaining aquifer lifetimes at various locations in the two regions.

Konikow (2011) and other researchers have represented the large-scale withdrawal of groundwater as a transfer of mass from continental lands to the ocean, resulting in sea-level rises. He estimated that from 1900 to 2008 global groundwater depletion was

Figure 2. Crop-specific contribution to groundwater depletion worldwide in 2010. The pie charts show fractions of groundwater depletion for irrigation (GWD) of major crops by country, and their sizes indicate total GWD volume. The background map shows groundwater stress index (corresponding to overexploitation when larger than one) of major aquifers. Some countries have overexploited aquifers, but no pie chart is shown because groundwater use is not primarily related to irrigation (Dalin et al. 2017).



about 4500 km^3 for a sea-level rise of 13 mm, just over 6 percent of the total sea-level rise.

Current concerns about future climate variability often anticipate increased local groundwater withdrawals with even faster aquifer depletion. Gorelick and Zheng (2014) provided a thoughtful discussion of multiple aspects of groundwater vulnerability relative to conflict, ecosystems, hazards, food security, human health, and energy resources. Combinations of these concerns are site-specific, and proper selection of simulation tools and policy constraints is necessary to plan sustainable groundwater management approaches.

Pumping Intensity

Much of the historical aquifer depletion has been caused by intense regional pumping for crop irrigation. Municipal and industrial wellfields can also cause depletion, but their lateral extents are usually smaller than the irrigated farming areas nearby. Figure 2 from Dalin et al. (2017) is an interesting representation of the crop mixes in 26 depleting aquifers around the world. Their groundwater stress index is greater than 1 when the aquifer is overexploited, and less than 1 when the recharge exceeds withdrawals across the aquifer.

Irrigation is one of the risk management choices available to agricultural producers when on-site groundwater is available and local rainfall is judged sufficient for acceptable yields. The producers also choose which crops are planted on how much area, which cultivation practices are used, and which fertilizers and pesticides are appropriate. Historically, the producing landowners and pertinent government or management agencies have used the groundwater for crop selections to please target markets, whether for profit or economic independence (reducing imports). These concerns directly relate to Tony Allan's famous virtual water concept (Allan 2011).

Municipalities often look to groundwater sources as groundwater typically requires less treatment than surface water and may be in closer proximity. Industrial and mining consumption (including oil and gas production) can be locally intense. Municipalities and industries tend to plan water use for decades for financial purposes. Oil and gas producers tend to use groundwater for relatively short times at each location.

Preventing Depletion

Annual recharge rates are often much smaller than the annual irrigation rates used for most irrigated crops. One extreme alternative to prevent depletion is to limit the locations of farms with high water needs to places with ample and timely rainfall. Of course, complementary soil, weather, and cultivation conditions would also be required by each crop.

Another alternative is to reduce the amount of area planted with crops that require irrigation, such as irrigating corn on one-fourth of the farm and raising rainfed sorghum or milo on the remaining area. Careful monitoring and maintenance of modern irrigation application systems can increase application efficiency to over 95 percent, but in some areas mechanized irrigation systems have led to increased water applications.

Irrigators also can choose to irrigate their crop at rates well below the level required for maximum yield. Crop genetics corporations have developed new plant strains that generate larger yields under rainfed conditions, and producers have adopted them, especially for cotton in the High Plains.

While these alternatives sound reasonable, implementation can vary due to the differences in groundwater ownership and production regulations. If the federal or state government owns the water and approves permits for annual pumping amounts, the stated alternatives could be applied with proper scientific planning of crop/soil/cultivation combinations and good weather predictions. Bringing all these items together has rarely occurred to date.

If the landowners own the groundwater beneath, the uncertainties of the crop markets encourage many to maximize short-term income by irrigating as much as possible, then converting the farm to rainfed or dryland crops later. Groundwater management districts can encourage or impose pumping restrictions, but to date most of the restrictions in place do not constrain the withdrawals enough to prevent long-term depletion. For example, most of the groundwater conservation districts in Texas have production limits that are best termed as managed depletion.

Municipalities and industries are able to pay much more for production and treatment of their water supplies than agricultural producers. Alternative surface water sources may exist. Aggressive conservation practices, such as those in El Paso and San Antonio, Texas, can reduce the amount of supply needed per capita, but that reduction might be overcome by the growing number of customers. Some have turned to deeper brackish aquifers, coastal seawater, or treated wastewater effluent reuse as the shallower fresh groundwater aquifers have declined, enduring the added treatment costs.

Aquifer storage and recovery (ASR) has been in use in El Paso since the 1980s and is gaining more adherents in the southwestern United States, such as Arizona, New Mexico, and Texas. Successful ASR requires careful management of the groundwater flow hydraulics and water quality blending. Water legal experts are still struggling with the ownership rights for ASR entities.

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Groundwater: State of the Science and Practice

Groundwater Management

By Robert E. Mace, Ph.D., PG

Merriam-Webster defines managing as handling or directing with a degree of skill, so it follows that managing groundwater use must mean handling or directing groundwater use with a degree of skill. A range of critical skills is needed to manage groundwater—including political, organizational, and communication skills. In addition, groundwater science—the systematic study of aquifers through observations, experiments, and modeling—provides key information to support decisions on managing groundwater and assessing whether management goals are being met.

The application of science to managing groundwater is broad—almost anything done to better understand a groundwater system is useful for management. For example, using isotopes or other natural or artificial tracers to better understand the timing and location of recharge; water-level measurements to understand flowpaths, cross-formational flow, and water-level dynamics; aquifers tests to understand responses to pumping; chemistry to understand water quality; and numerical models to project aquifer response to changes in pumping and recharge.

The purpose of this chapter is to describe how groundwater science can inform the management of groundwater resources. This discussion is not intended to be comprehensive—I would need the entire book to do that. Instead, this is an introduction on how groundwater science can be employed to assist in groundwater management.

To facilitate the discussion, I decided to start with the assumption there is no management and then sequentially, level by level, build a scientific program from the ground up (or actually, from the ground down!) to support management. I use this approach to discuss various scientific tools that can assist in understanding and managing aquifers. However, each aquifer and the realities of its sociopolitical overlay requires its own path best determined by the appropriate local, national, and perhaps international experts.

Regardless of how an aquifer is managed or, for that manner, not managed, science is helpful in understanding what is happening in an aquifer and why. In unmanaged areas, there may already be baseline science available from governmental and academic studies.

If little to no information is available, then the first order of business should be a well survey to assess existing boreholes in the area, ideally with information on location, land-surface elevation, well depth, depth to water, use, volume of use (generally estimated), conductivity, and temperature.

Depending on the local requirements for well drillers, a groundwater manager may be able to acquire more details on the well, including screen depth and length, well yield, and a record of the geologic materials encountered when sinking the well. In addition, springs should be surveyed as well.

For the groundwater manager, a well survey is invaluable. The survey tells a manager where the wells are, how close they are, who's using them for what, and where wells might be interfering with each other and springs and rivers. It's also an opportunity to develop a relationship with well owners and the public.

With a well and spring survey in place, a groundwater manager will then be able to design a monitoring network where groundwater levels can be measured annually at around the same time each year, preferably in a part of the year when pumping is at a minimum (for example, when there is minimal irrigation use). Over time, this information will tell a manager what is happening to water levels in their area. Are they rising? Declining? Staying static? A subset of, ideally, unpumped wells could also be measured more frequently—at least monthly—to assess seasonal changes.

Employing the latest technology allows real-time monitoring of water levels that can be posted to the Internet for all to see and use. Third-party vendors are now offering services to well owners to remotely monitor water levels in real-time.

A higher level of water-level monitoring could involve the groundwater manager drilling dedicated monitoring wells or packed wells that allow monitoring at different elevations in the borehole. A manager-owned monitor well avoids uncertainties that may be encountered when working with privately owned wells, such as loss of access with change of ownership or a decision to use the well. Remote sensing tools are being developed using NASA's GRACE (Gravity Recovery and Climate Experiment) to assess changes in groundwater volumes, assuming other changes, such as in soil moisture, can be accurately factored out.

A well and spring survey also allows for systematic water-quality sampling across a managed area. Unlike water-level monitoring, wells sampled for water quality do not have to be unused (in fact, testing a well in use may be preferred since it doesn't require the installation and use of well-development equipment). Expanding beyond conductivity and temperature, wells can be tested for anions and cations.

If there are contamination concerns, then these wells could also be tested for an array of contaminants. A well and spring survey further allows for the development of a well-testing program where aquifer tests provide information on the permeability of the aquifer. A groundwater manager can use this sampling and testing information to assess where there are water quality concerns for crops, industry, and human consumption and where wells can support certain uses of groundwater.

If the local geology is not known, then it is important to map the geology, including the subsurface geology. Hopefully enough information was collected from existing wells to be useful when mapping the subsurface.

Any oil and gas exploration in the area may be invaluable to such studies since companies seeking subsurface petroleum often collect a suite of downhole geophysical data that hopefully started shallow enough to capture the local aquifers. The drilling of deep boreholes to explore for oil and gas often requires a source of water for drilling operations; therefore, many explorers will drill a nearby groundwater well for supply. These wells generally have good information on them that can be invaluable for understanding local groundwater resources.

Absent any reliable information on the subsurface, a groundwater manager may be required to drill their own boreholes to explore the subsurface. The aforementioned manager-drilled monitor well could also be used to more fully explore the subsurface.

Geophysics can also be employed to explore the subsurface. A variety of methods with different strengths and weaknesses are available for use, including electrical resistivity, seismic, ground-penetrating radar, magnetotellurics, electromagnetics, nuclear magnetic resonance, and microgravity. Geophysical surveys can be cost-prohibitive for regional studies but extremely useful for local-scale, shallow aquifers and for exploring certain features of interest in regional aquifers (such as a fault). A good geologic understanding helps a groundwater manager understand well interference and is critical to developing additional scientific tools for management.

With a geological understanding in place, wells can then be associated with the geology. Once associated, water-level elevations can then be contoured providing information—assuming the network is dense enough—on where groundwater is coming from, where it is going, and how it is interacting with surface-water bodies. Data collected over time will also show how and where regional water levels are changing.

With an understanding of where water is coming from and where it is going, geochemists can then be employed to assess the timing and location of recharge as well as cross-formational flow and chemical development of groundwaters as they flow through the aquifer. The geochemists can confirm and inform the conceptual understanding of how water is moving in, through, and out of an aquifer. Gain-loss studies on rivers and streams that cross the recharge and discharge zones of the aquifer also assist in understanding recharge and natural discharge.

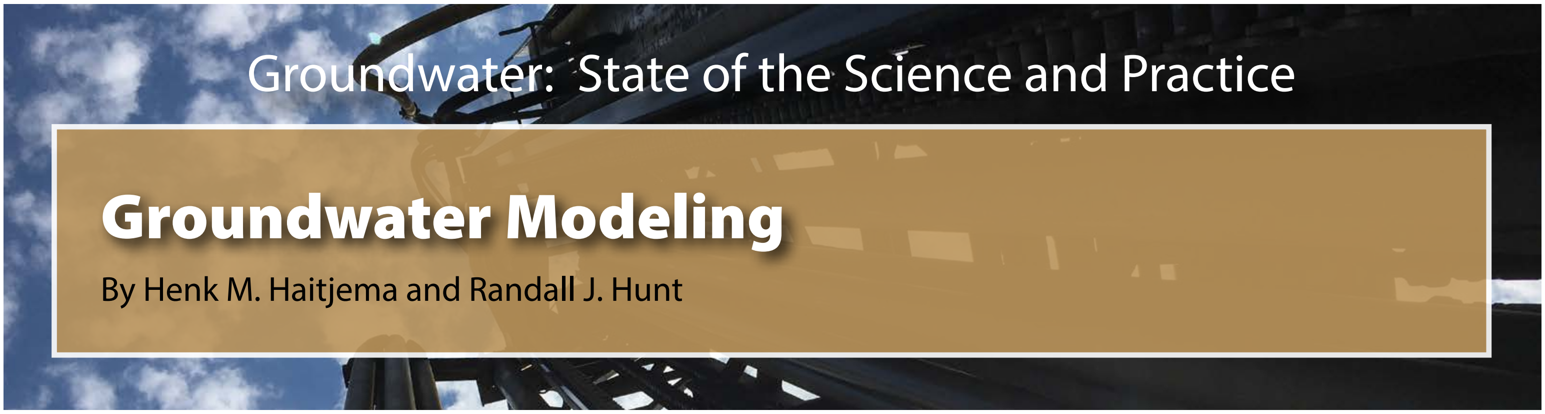
With all the previous information on water levels, hydraulic properties, recharge, and discharge and a solid conceptual idea of how the aquifer works, a numerical groundwater flow model can be developed.

A groundwater model is particularly valuable to groundwater managers in that they can project how future levels of recharge and pumping might affect the aquifer. With a calibrated model in place, decision support tools can assist policymakers, managers, and the public in making critical decisions on managing their groundwater resources. A model can also be useful in assessing aquifer storage and recovery, enhancing recharge, and managing groundwater conjunctively with surface water (or other groundwater systems).

In addition to its management benefits, a model also has immense scientific value in that it tests the conceptual model and identifies gaps in knowledge that future data collection can strive to fill and thus improve the model.

A challenge to hydrogeologists is that policymakers, managers, and the public demand (or assume) a level of precision and accuracy that is generally not achievable in the science. Therefore, it is critical to convey the appropriate level of uncertainty when discussing the application of science to any management aspect of the aquifer. Because of this uncertainty, it is advisable that groundwater managers employ adaptive management in their regulations. Adaptive management allows management goals, rules, and permits to respond to reality and future improvements in understanding the aquifer.

Groundwater managers also need to appreciate that collecting groundwater data, improving the understanding of the aquifer, and improving groundwater models are ongoing business expenses. Groundwater management and groundwater science need to work together to achieve groundwater goals.



Groundwater Modeling

By Henk M. Haitjema and Randall J. Hunt

Introduction

The state of the science and practice in groundwater modeling brings to mind highly sophisticated computer models that are running in parallel on many multi-processor machines. These models are expected to incorporate many different processes of both saturated and unsaturated groundwater flow and transport and possibly the media to which it connects, like surface waters and the atmosphere. We are increasingly aware we cannot study groundwater flow in isolation if we are to make useful predictions of, for instance, the impacts of climate change on the groundwater regime. We have come a long way.

Today we are no longer limited to equations for flow toward a well, perhaps near an infinitely long straight canal (method of images), to sandbox models in the laboratory, or to simple steady state models of flow in a single aquifer. We now have computer models that solve groundwater flow and transport in multi-aquifer settings under transient conditions and with a user-friendly graphical user interface that allows widespread use. Additionally, multi-media models are now leaving the research environment and becoming available to mainstream consultants. So in that sense the *science* of groundwater modeling has matured.

The *practice* of groundwater modeling, however, has also matured. We have come to realize that model output, being a necessary simplification of an unknowably complex natural world, has inherent limitations. That is, a model of reality is not reality itself. There is uncertainty associated with all facets of our model—parameterization, aquifer geometry and discretization, boundary conditions, and future hydrologic drivers such as future pumping regimes and climates. Today a model is now more appropriately seen as a tool that provides a quantitative framework to make supportable forecasts rather than an oracle that gives us all the answers.

In this chapter we set out to briefly review the state of the science and practice in modeling. In doing so, we augment existing assessments from the journal *Groundwater* (e.g., Hunt and Zheng 2012; Langevin and Panday 2012; Molz 2017a,b; White 2017), specifically in terms of modeling approach. An effective modeling approach is critical. If a modeler does not decompose the societal problem correctly, the model will not be fit-for-purpose, no matter how sophisticated the code's capabilities. Moreover, capabilities of codes will be ever improving; good modeling practices have a timelessness that is more robust.

How best to decompose the problem and provide models that are accepted? We lay out here some approaches for today's applied groundwater modeling. Specifically, we suggest: (1) a step-wise modeling process; (2) including a two-dimensional areal model within this process; (3) keeping abreast of industry standards; and (4) ways to increase acceptance of the models we produce.

What is a step-wise approach?

In a nutshell, a step-wise approach increases insights into the groundwater flow, or transport, problem because we add complexity as we note deficiencies in simpler approaches, and add more complexity (and more detail) only when needed to address the deficiencies identified.

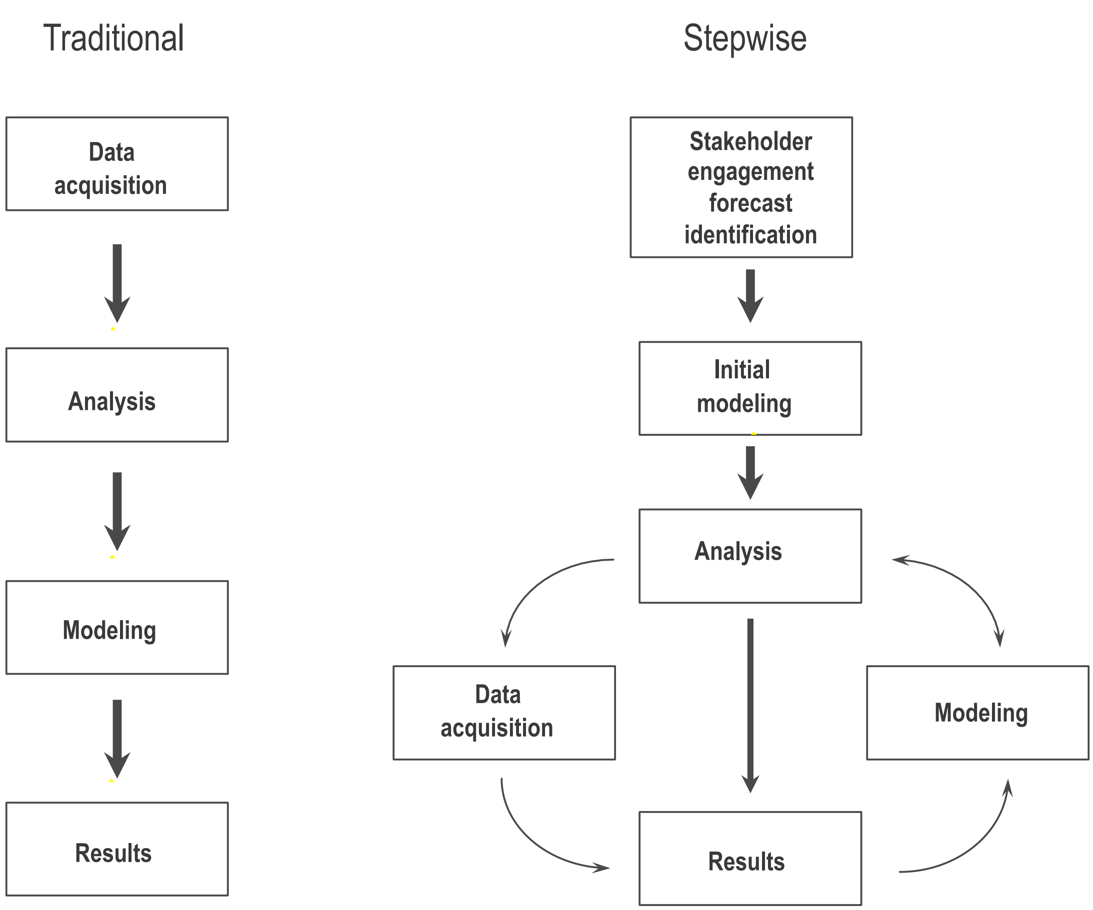
Even if the end goal of our project is to develop a state-of-the-art multi-media model, much is to be gained from some initial scoping or screening models, or even hand calculations, that help to build insight and ultimately confidence in the final modeling results. Moreover, such a focus will facilitate final models that are on time, within budget, and effectively answer the questions asked of the modeler.

Traditional groundwater modeling studies followed a linear workflow as shown in Figure 1. Since the data acquisition step, possibly including a field campaign, was completed well before modeling results became available, the modeler frequently found out (too late) data were missing in critical areas and redundant data had been collected in other areas. This process was not only frustrating but also costly and incomplete, which in turn helped to give groundwater flow modeling a bad reputation in some circles.

The step-wise iterative modeling approach shown on the right in Figure 1, in contrast, turns the process on its head. Instead of starting with data acquisition of what we think is needed where we think it is needed, we start with stakeholder engagement whereby the purpose of the model is articulated and the specific model forecasts of interest identified.

A first cut at the system is simulated in (initial) modeling, whereby the forecasts of interest are given equal attention as existing observations from the system. This "forecast first" focus in the earliest stages of modeling is vital for effective modeling (White 2017), regardless of step-wise or traditional. The model that is best fit-for-purpose is one that provides

Figure 1. Traditional modeling (left) and step-wise modeling (right). (Modified from Haitjema 1995).



forecasts of what we don't know with the least uncertainty, which may not be the model that simply best reproduces what we already know (as expressed by the calibration data).

Of course, initial models require some basic information about the groundwater flow system, but today we have sufficient existing data for most groundwater systems to get "in the ballpark" and our models should not require a field campaign to get started. The initial modeling can be a few simple hand calculations of water balance or a simple one-layer model to gain insight in possible groundwater flow directions. Since most aquifer parameters will not yet be well constrained at this point, different values may be used to establish sensitivity to our forecasts of interest as well as our observations from the system.

This process guides data acquisition so that we only pursue relevant data in relevant areas as suggested by the initial modeling results and analyses. Employing parameter estimation software at this stage can provide a quantitative framework to assess the worth of potential future data collection (e.g., Fienen et al. 2011). Subsequent modeling steps involve adding complexity to the model as more data become available and acquiring supplemental data as new shortcomings in our understanding are identified.

The process converges when the modeling results do not significantly change in subsequent iterations. This process takes full advantage of early model insights to direct effort where it is most needed. Moreover, it helps ensure maximum fit-for-purpose at a minimum cost since little or no unnecessary data are being acquired and no unnecessary model complexity is introduced.

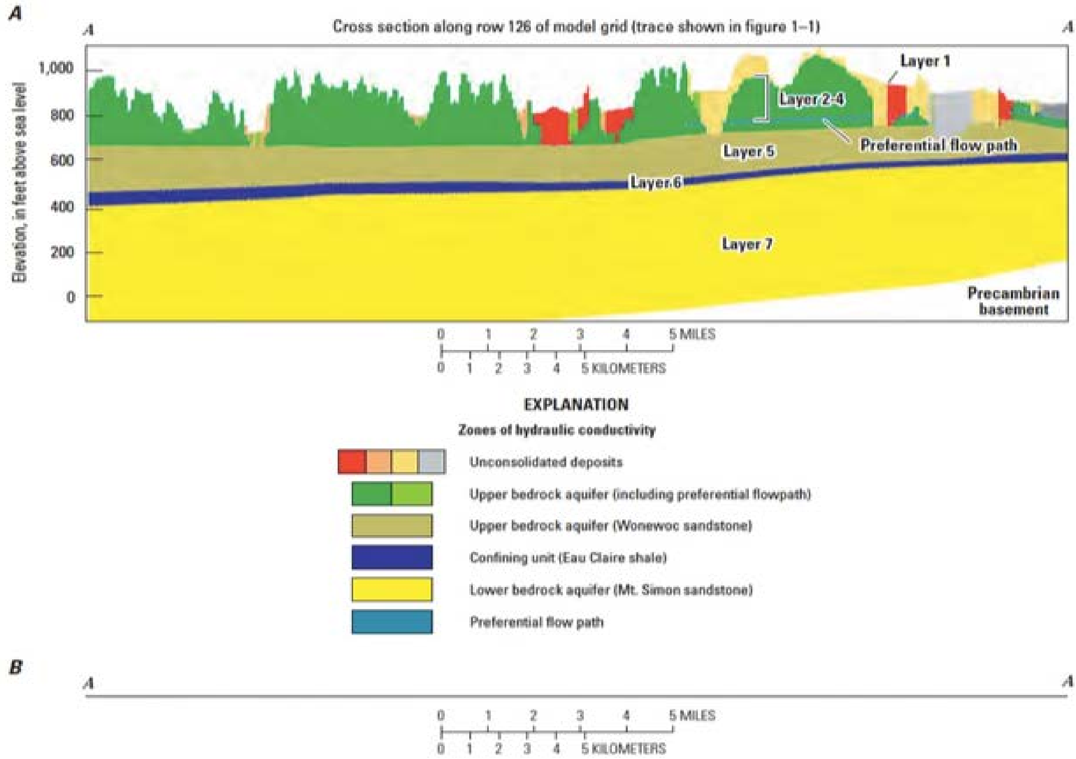
Putting the modeling odds in your favor with Dupuit-Forchheimer

Highly dimensional models—whether in space, time, or medium—require more model input, have longer model runtimes, and are characterized by more model instability. To simply quality assure the inputs can be time consuming, and problems with input, or translation of the conceptual model, can be difficult to identify in isolation. Early models developed in the step-wise approach, however, can provide a reality check on more complex multi-dimensional models. That is, large disagreements in results—such as direction of flow or water balance—can direct the modeler's eyes to problematic input and/or conceptualization.

One of the easiest ways to obtain a reality check for more complex dimensional models is to include a two-dimensional areal flow model (or "Dupuit-Forchheimer" or DF model) in a step-wise modeling workflow. Why focus on this particular model design? Because the art of modeling is keeping what you need for the problem and omitting what is not (Anderson et al. 2015). For most groundwater problems and aquifer dimensions, downsides of simplifying the vertical dimension are smaller than the upside of quickly gaining the insights on the important characteristics of the system of interest, such as constraining flow into and out of the area of interest (e.g., Hunt et al. 1998).

The appropriateness of focusing on horizontal groundwater flow is perhaps most easily seen by plotting up a cross section of an area of interest *without vertical exaggeration* (e.g., Figure 2). Note that without vertical exaggeration, the cross section appears to be a straight line (Figure 2B). Harkening back to the days of simulating groundwater flow with analog models, imagine the path of a molecule of water—will it see more resistance to flow in the vertical dimension when the distance traveled in the horizontal is typically thousands to hundreds of thousands times longer?

Figure 2. A comparison of a 100X vertically exaggerated cross section of a groundwater system (A) and the same cross section removing vertical exaggeration (B); the latter reflects the true dimensions of the system in the field (from Hunt et al. 2016).



The history of DF models is rich, and yet misconceptions still arise, such as limiting thinking of them to solely "horizontal flow models." In the editorial "Horizontal flow models that are not" (Haitjema 2016), it is explained that DF models do include vertical flow, albeit in an approximate manner. That is, since such two-dimensional areal models often include recharge at the aquifer top (and sometimes leakage through the aquifer bottom), vertical flow must be present.

Paradoxically, DF models have been shown to successfully simulate such flow systems; how can this be? Kirkham (1967) explained the paradox by demonstrating we are not ignoring vertical flow, but only the *resistance* to vertical flow. Indeed, Strack (1984) presented a complete theory of three-dimensional pathlines in DF models, where horizontal flow is calculated using Darcy's law and vertical flow is approximated using mass balance. In fact, the widely used multi-layer MODFLOW models are nothing more than a stack of DF models in which vertical flow is obtained from water balance. On a regional scale (where the horizontal travel distance is much larger than the vertical distance), we know DF models produce potentiometric head surfaces and three-dimensional streamlines that are nearly indistinguishable from fully three-dimensional models (e.g., Haitjema 1987).

Consequently, DF models are excellent early screening/scoping models even when the endpoint is a more comprehensive and sophisticated groundwater model. Having such a representation of your system in hand will help increase the odds of a successful model that is on time and within budget.

Going with the flow: MODFLOW and MT3D

Although there are numerous modeling techniques such as finite differences, finite elements, and analytic elements, the most widely used groundwater model is MODFLOW, which is based on the finite difference method. MODFLOW (McDonald and Harbaugh 1983) is a modular code developed and maintained by the United States Geological Survey (USGS). Although around a long time, its modular code structure facilitates ongoing development with the addition of ever more sophisticated hydrological processes.

Starting as only a groundwater flow model, numerous "packages" were added over the years, such as ever more powerful matrix solution procedures, seawater intrusion, conjunctive groundwater flow and streamflow, flow in the unsaturated zone, and evapotranspiration. With the release of MODFLOW-USG (2013) and MODFLOW 6 (2017), the finite difference limitation of a rectangular grid was overcome using an unstructured grid, which allows the use of a variety of cell shapes to improve the representation of streams and wells, for instance, in the model grid.

In September of 2016, the USGS released its own version of the popular MT3D contaminant transport model, which uses a MODFLOW-supplied flow field. MT3D was first released in 1990 (see also Zheng 2009). One reason MODFLOW and MT3D have become generally accepted as the industry norm is that they are freely available and their source code is available. Such accessibility and openness fosters a community of users and developers, as well as more widespread code testing and transparency of how results are obtained.

Yet, some applied modeling problems will benefit from moving beyond the industry standard. Indeed, even the leading international conference on MODFLOW modeling in the United States is called "MODFLOW and More" in recognition that there is no one-size-fits-all approach to groundwater modeling. The journal *Groundwater* featured a special section devoted to the 2017 MODFLOW and More conference in issue 4 (July–August) of 2018. Therein one can see a snapshot of what is new in both the MODFLOW realm, as well as the "more" that reflects the world beyond MODFLOW.

Increasing acceptance of the models we build

Gone are the days where an "expert" can provide a model's output and have it accepted by all parties without question. Today's models reside at the interface of decision-making, focusing discussion, providing quantitative frameworks for "what-ifs" that may be suggested, and enhancing extraction of stakeholders' information about the system. In this way stakeholders are participants in the process as modeling is being performed (Bots and Daalen 2008).

Within this participatory modeling framework, it is likely that the ensemble of possible models considered includes those that fit a modeler's conception of the natural world, but also those that stakeholders believe are representative of their system. Such "advocacy driven" models (Ferré 2017) are important not only to ensure the broadest net is cast, but also to facilitate identification of data that could be collected that discriminates between potential model endpoints. At the same time, by their participation in the process, a stakeholder has more time to understand what lies within a model, which increases acceptance of the final model result and its associated forecasts. Interestingly, this ensemble approach harkens back to Chamberlin's (1890) method of multiple working hypotheses, and highlights how modeling is, in fact, part of the scientific method being used to answer questions relevant to society.

Finally, it is now widely accepted that model outputs cannot be considered "the answer." Rather, all models have uncertainty surrounding their inputs and outputs. Indeed, an entire chapter is devoted to uncertainty analysis in the applied groundwater modeling textbook of Anderson et al. (2015). Therefore, modeling best practices recognize the utility of providing an estimate of uncertainty along with model forecasts for decision-making. At the same time, it is recognized that just as there cannot be one "true model" of the unknowably complex natural world, there cannot be a "true" estimate of a model's forecast uncertainty (Hunt 2017).

Therefore, which method was used to calculate forecast uncertainty is less important than the simple act of stating the uncertainty the modeler feels is representative. Often even simply relating the modeler's expected uncertainty around the forecast, based solely on their experience running and calibrating the model, increases acceptance of their model, which in turn adds value to the decision-making process.

The suggestions above reflect strategies for successful and cost-effective modeling in today's legal, technical, and societal arenas. However, what has not changed is that the skill and expertise of the modeler are what ultimately facilitates credibility and acceptance. Thus, as pointed out in Hunt and Zheng (2012), "with great power comes great responsibilities." When all is said and done, new capabilities and advances in the science and practice of groundwater modeling can only serve to augment and empower the underlying *hydrosense* already present within the modeler.

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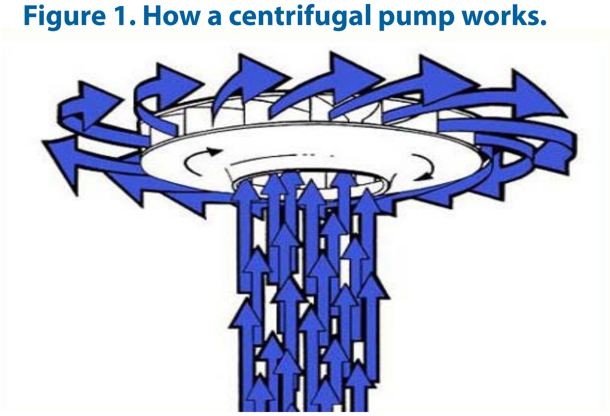
Groundwater Pump Systems

By Dave Kill

Centrifugal pumps are the most common type of pump used in groundwater pumping. This is the case if it is a single stage end suction pump such as a jet pump or a vertical multi-stage that is installed directly in the well. A vertical multi-stage pump can pump from much greater depths as additional stages provide more head capability.

Figure 1. How a centrifugal pump works.

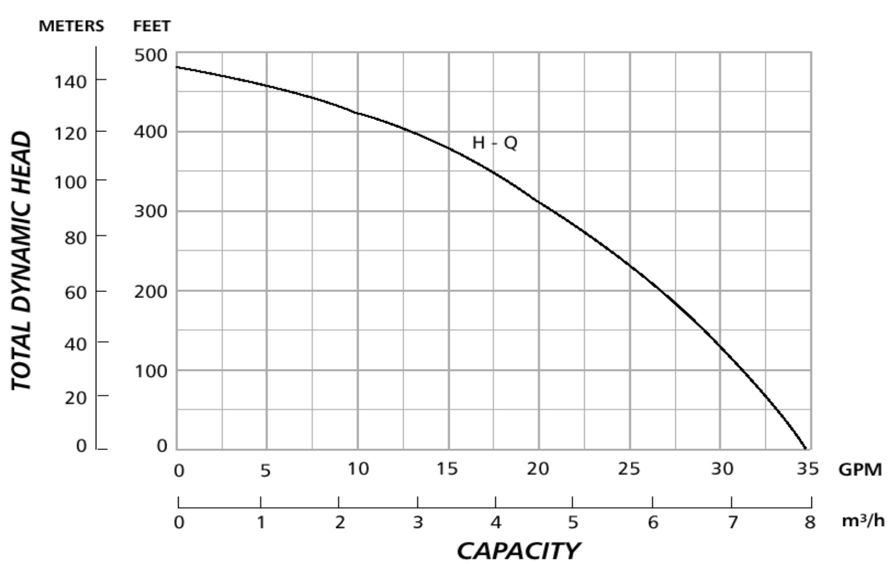


Theory of a Centrifugal Pump

Fluid enters the eye of the wheel (impeller) and is forced to the outside via centrifugal force (Figure 1). The speed vector at the outer edge of the impeller determines how much energy or head that is imparted to the fluid. The speed vector is a function of rotating speed and diameter of the impeller. This energy is captured in a housing (bowl) and either directed to the pump discharge or the next impeller. Changing the rotating speed is used to change the performance of the impeller, which is the advent of constant pressure water systems that have a varying capacity requirement.

The performance of a centrifugal pump is denoted by a H-Q curve as shown below. It relates H total dynamic head (TDH) and Q capacity (GPM). The pump always operates on this curve with the controlling factor being the TDH or pressure head the pump is pumping against.

Figure 2. Performance of a centrifugal pump denoted by a H-Q curve.



Groundwater pump selection is done by determining what TDH is required for the needed water system capacity. Determination of TDH is done by adding lift in the well (PWL—pumping water level) plus friction loss in the piping system and pressure required in the water system. Units for these three factors must be in feet of head in order to be directly additive.

The vertical multi-stage submersible pump as shown below is usually classified as low capacity or high capacity. Low capacity is generally up to 100 GPM as the standard 4-inch-diameter submersible pump can achieve this capacity. This 4-inch-diameter pump will fit in a 4-inch well casing as the pump diameter is less than 4 inches, including electric cable guard. Greater than 100 GPM requires a larger diameter submersible pump, so it is generally considered high capacity and requires a larger diameter well.

Low capacity submersible pumps are typically fitted with a flat or pancake impeller as shows in Figure 3. Materials of construction of this flat impeller is either thermoplastic or stainless steel. High capacity impellers are the turbine type.

Figure 3. Vertical multi-stage submersible pump.

Figure 4. Flat, pancake impeller for low-capacity submersible pump.

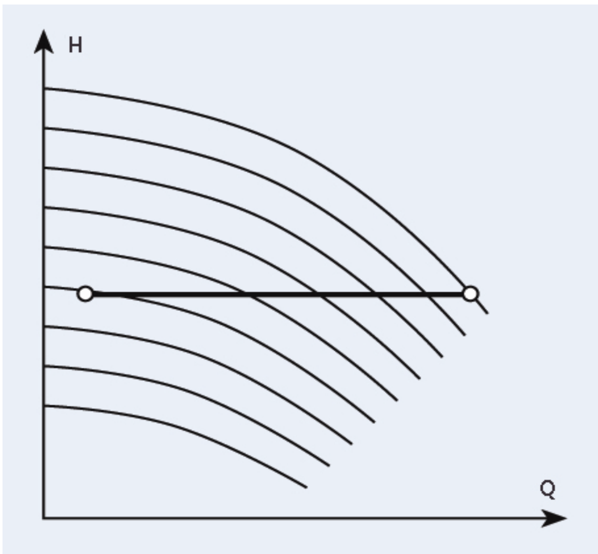
In groundwater pumps the construction material of this impeller is bronze or stainless steel.

The main difference between the two impeller types is the hydraulic efficiency is much higher for the turbine type. It is not unusual to have hydraulic efficiency in the mid-80% range for the turbine type impeller. Flat impellers are rarely above the mid-50% range for hydraulic efficiency.

All groundwater pumping centrifugal pumps are driven by electric motors whether they are a single-stage jet pump or a multi-stage pump. Submersible pump electric motors are specifically designed to be placed in the water well below the water level. High capacity turbine pumps may also be driven with an above-ground motor and a long drive shaft from the motor to the pump assembly.

The motors are either 2 pole (3500 rpm) or 4 pole (1750 rpm) speed at 60 Hz. Either single phase or three phase motors are available, with single phase limited to 10 HP and below. Voltages are commonly 115, 230, or 460 volts. Other voltages and speeds are available in very large HP motors.

Figure 5. With the use of a variable frequency drive (VFD) the speed of the electric motor can be changed, thus changing the centrifugal pump performance. Changing the speed creates many different H-Q curves as shown below.



Each of the curves below the top one is a different rotating speed of the pump, with the top one being the 2 pole (3500 rpm) speed. All lower speed curves are parallel to the initial full speed curve. This is a characteristic of centrifugal pumps that makes them very conducive to the VFD application. The horizontal line shown on the multiple curves is the constant pressure no matter the flow rate required in the water system.

Another advantage of the VFD controlled pump is the reduction of brake horsepower (BHP) as the pump rotation is lowered. This is defined by the centrifugal wheel affinity law. BHP changes as the cube of the speed change. The result is lower BHP and reduced electric cost.

Figure 6. Relate performance changes to change in speed.

$$Q_2 = Q_1 \left(\frac{N_2}{N_1}\right)$$

$$H_2 = H_1 \left(\frac{N_2}{N_1}\right)^2$$

$$bhp_2 = bhp_1 \left(\frac{N_2}{N_1}\right)^3$$

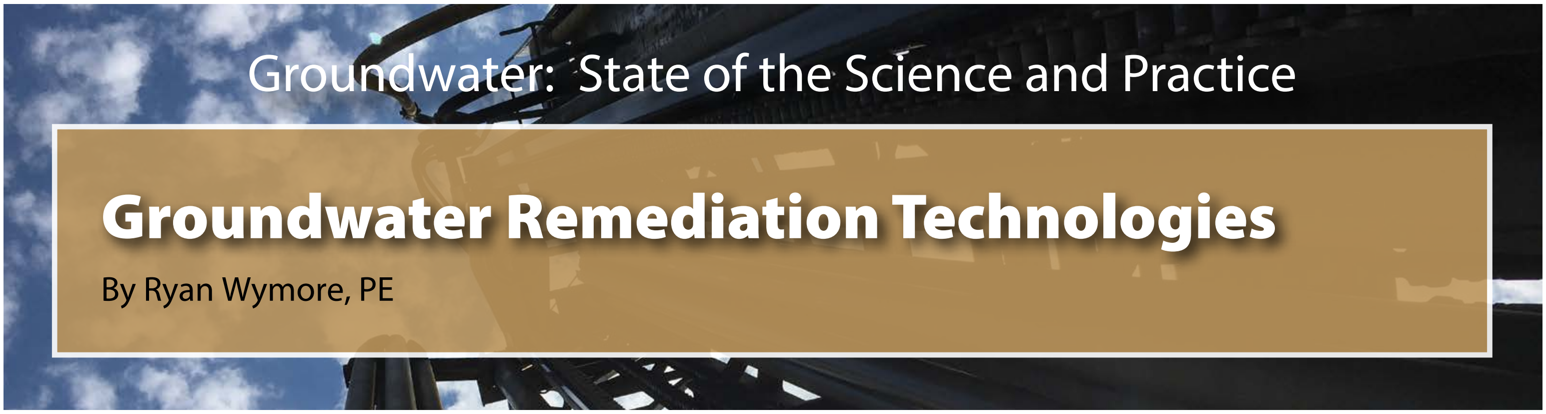
The following example shows how much the BHP is lowered by lowering the speed.

Reduce speed 20% (60 Hz to 48 Hz)
reduces HP by 49%.

48/60 = (.8)(.8)(.8) = .51

This is a significant reduction in operating cost.

In selecting a pump for groundwater pumping, anticipation of the pumping conditions is most important. In other words, what will be the maximum capacity required and what will be the lowest expected pumping water level in the well? Quite often these conditions are not known, and they can be variable depending on the water system operation. Well design and completion can also have an influence on the pump selection, particularly well diameter and well depth.



Groundwater Remediation Technologies

By Ryan Wymore, PE

Groundwater remediation is the process that is used to treat contaminated groundwater by removing the pollutants and/or converting them into harmless products. Many technologies have been applied to remediate groundwater over the past 30-40 years.

Early remediation efforts relied heavily on pump and treat, where contaminated groundwater is pumped and treated above ground. The treated water can be reinjected into the same aquifer or discharged to surface water or to the area's public sewer system. In fact, pump and treat has been selected or is still being used at more than 800 Superfund sites across the country (EPA 2012).

Today, in situ remediation technologies (i.e., technologies that treat groundwater "in place" without first pumping groundwater to the surface) are commonly implemented at contaminated groundwater sites. These are the focus of this chapter.

In Situ Remediation Technologies

Over the past 20 years, there has been a proliferation of in situ remediation technologies that rely on biological, chemical, and thermal mechanisms. Numerous guidance documents, fact sheets, practice and design manuals, etc. are available for many of these technologies. As such, a brief description of each of these technologies is provided.

In Situ Bioremediation

In situ bioremediation uses microbes to degrade contaminants within an aquifer. Often this requires addition of an amendment to stimulate growth of certain bacteria. Microbes gain energy through respiration reactions where they consume a food source (also called an electron donor) and "breathe" an electron acceptor (e.g., oxygen). Microbes degrade some contaminants by using them as the electron donor, while they degrade other contaminants by using them as the electron acceptor.

To implement in situ bioremediation as a remediation technology, amendments are added to the subsurface to stimulate the targeted component(s) of microbial metabolism. For contaminants that are used as electron donors (such as petroleum hydrocarbons), the added amendment is oxygen because the degradation reactions are usually aerobic. Several options are available for adding oxygen, including direct injection of air or oxygen, dissolving oxygen in extracted water and reinjecting it, and various oxygen releasing compounds.

For contaminants that are used as electron acceptors, an electron donor is added as a food source for microbes to stimulate contaminant degradation. The most common type of bioremediation where contaminants are used as electron acceptors is a process known as anaerobic reductive dechlorination, which is applicable for chlorinated solvents. Many types of electron donors are available for reductive dechlorination.

Soluble amendments such as lactate or molasses can be used; these amendments are relatively inexpensive and can stimulate rapid degradation, but they have a short longevity in the subsurface and therefore are known as fast release donors. Other amendments are designed to stimulate slower degradation but have greater longevity in the subsurface and are therefore known as slow release donors; an example is vegetable or soybean oil-based amendments.

For all forms of bioremediation, various nutrients can be added, which sometimes can improve degradation efficiency and rates. Also, in some cases, the bacteria that are needed to degrade the target contaminants may not be present at a site. For some contaminants such as chlorinated solvents, bacterial cultures are commercially available that can be added to a site's groundwater along with the appropriate amendment; this process is called bioaugmentation. One recent resource produced by SERDP/ESTCP is available that provides guidance for bioaugmentation (ESTCP 2012).

In Situ Chemical Oxidation

In situ chemical oxidation (ISCO) involves injecting chemical amendments to stimulate the complete oxidation of contaminants, resulting in their breakdown to carbon dioxide, water, and for chlorinated solvents, inorganic chloride. ISCO technology is based on the oxidative power of the chemicals that are added. Some oxidants are stronger than others and therefore can degrade a wider array of contaminants and/or degrade some contaminants faster than other oxidants.

ISCO can be used to degrade many types of contaminants including solvents, metals, fuel/petroleum constituents, pesticides, and munitions constituents. The type of oxidant used is largely determined by the contaminant. Common oxidants include hydrogen peroxide, permanganate, ozone, and persulfate.

An advantage of ISCO compared to bioremediation is that treatment can occur more quickly, as ISCO is generally considered to be a more aggressive technology. However, because of this, the oxidants can be short-lived in the subsurface, which can result in contaminant "rebound"—a phenomenon where contaminant concentrations decrease temporarily in response to treatment but then increase after some length of time after the amendment is consumed. This can be overcome through proper ISCO design and implementation.

In Situ Chemical Reduction

In situ chemical reduction (ISCR), like in situ chemical oxidation, relies on adding amendments to the subsurface to stimulate chemical degradation of contaminants, but unlike in situ chemical oxidation, the resulting degradation process is anaerobic reduction rather than oxidation. One common ISCR amendment is zero valent iron (ZVI), which is added to the subsurface directly as a microscale or nanoscale solid or is injected as a slurry or a liquid suspension.

ISCR is commonly used for chlorinated solvents but can also be used for some metals such as chromium. It is generally not effective for contaminants that can be oxidized, such as petroleum hydrocarbons.

Because ISCR and reductive dechlorination require similar aquifer conditions, they are commonly applied together, in some cases using amendments that combine ZVI and electron donors. It is now known that under anaerobic conditions that are created during reductive dechlorination of chlorinated solvents, reactive iron minerals can form in the subsurface that themselves can abiotically degrade the contaminants via abiotic reduction. Therefore, ISCR can be implemented using various approaches that add the chemical reductants directly, or that use amendments that stimulate biological processes that then form the reactive iron minerals.

In Situ Thermal Remediation

In situ thermal remediation (ISTR) consists of heating subsurface soil and groundwater to facilitate contaminant destruction and volatilization, combined with contaminant extraction and treatment. ISTR has gained wide acceptance over the past 20 years as a source reduction technology that is effective for many contaminants. A recent Department of Defense document provides guidance on design and implementation of ISTR (DoD 2006).

Three common ISTR methods used today include electrical resistance/resistive heating, thermal conductive heating, and steam-enhanced extraction.

Electrical resistance/resistive heating (ERH) involves heating by the passage of electrical current through the subsurface using arrays of electrodes. The subsurface provides resistance to the applied electrical current, which results in increased groundwater temperature. ERH has been applied extensively for treatment of a wide variety of contaminants, especially chlorinated solvents. Although this is an efficient way to heat the subsurface, it is limited to the boiling point of water at 100 degrees Celsius since the conducting material (water) is removed once the water is vaporized above 100°C.

Thermal conductive heating (TCH), also often referred to as in situ thermal desorption, uses conduction heater wells to heat the subsurface. These heater wells are operated at high temperature (up to 800°C), which causes heat to propagate throughout the target treatment zone. TCH has been used for a variety of contaminants, including chlorinated solvents. Because temperatures can be increased to significantly greater than 100°C, additional destruction mechanisms are possible, including hydrolysis, oxidation, and pyrolysis. This also means that contaminants with boiling points higher than water can be remediated using TCH.

Steam-enhanced extraction (SEE) heats the subsurface using injection of steam. This heating results in mobilization and evaporation of contaminants toward the center of a treatment system for extraction. SEE, as with ERH, is generally limited to contaminants that volatilize at or below 100°C.

Monitored Natural Attenuation

Monitored natural attenuation (MNA) as a remedy refers to reliance on natural attenuation processes to achieve site-specific remediation objectives within a time frame that is reasonable compared to that offered by other more active methods (EPA 1999). These natural attenuation mechanisms include non-destructive processes like dispersion, dilution, sorption, and volatilization, as well as destructive processes like biodegradation, radioactive decay, and chemical or biological stabilization.

The U.S. EPA considers three lines of evidence before MNA can be accepted as the remedy for a site (EPA 1999):

- Historical groundwater data that demonstrate a clear and meaningful trend of decreasing contaminant mass and/or concentration over time
- Hydrogeologic/geochemical data that can indirectly demonstrate active attenuation mechanisms
- Data from field or microcosm studies that directly demonstrate the occurrence of a particular natural attenuation mechanism at a site.

MNA is commonly combined with active remedies. MNA may be appropriate for many types of contaminants, with its suitability depending on whether the EPA's lines of evidence can be established for a given site.

Strategies That Represent the State of the Practice

Because of the complex nature of many sites requiring remediation, straightforward implementation of individual remediation technologies is often not sufficient to achieve remedial goals. Nuanced application of combined remedial strategies and innovative monitoring and amendment delivery techniques can often improve remediation outcomes. While many such strategies are being used in the remediation industry today, three will be briefly discussed here.

First, because several in situ remediation technologies involve amendment delivery, selection of appropriate injection/delivery techniques is key. Many options are available for liquid amendments such as standard injection wells, injections using direct push technology (DPT) drilling, active recirculation (e.g., pumping groundwater to induce gradients and to control amendment delivery), and horizontal or angled wells.

Additional technologies are available for solid amendments such as trenching, and hydraulic and pneumatic fracturing. Ultimately, the selection of appropriate delivery technologies depends on the amendment selected, the site-specific conditions, and the remedial goals.

Secondly, advanced monitoring tools like environmental molecular diagnostics (EMDs) are being deployed more frequently to better understand degradation processes. EMDs are a group of advanced and emerging techniques used to analyze biological and chemical characteristics of environmental samples (ITRC 2013).

EMDs include several techniques that can assess and quantify specific microbes, enzyme systems, proteins, and/or metabolic functions, as well as other assays that can provide information about the overall microbial community. EMDs also include compound-specific isotope analysis (CSIA) and stable isotope probing, both of which can provide valuable information regarding degradation mechanisms and rates.

In addition to EMDs, alternate site management tools are becoming more common. One such approach is using mass flux/mass discharge to characterize sites and evaluate remedies. ITRC recently released a comprehensive guidance document describing principles of mass flux/mass discharge, as well as accepted tools and techniques for performing the measurements (ITRC 2010a).

Thirdly, in recent years there has been a shift to integrate or combine remedial technologies to maximize treatment efficiency. This integration may be temporal where technologies are connected in a logical sequence, or in a spatial manner where different technologies are used to address variable site conditions.

Some remedial technologies can be more easily combined than others, and certain combinations can offer synergies. One example is in situ thermal remediation and in situ bioremediation, where residual heat from a thermal remedy component can accelerate biological degradation reactions. The ITRC Integrated DNAPL Site Strategies document (ITRC 2010b)offers guidance on which technology combinations may be more favorable than others.

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Groundwater Sustainability

By William M. Alley

Introduction

Sustainability is a wide-ranging term that can be applied to almost all aspects of life on Earth, from the local to a global scale. The National Ground Water Association (NGWA) defines groundwater sustainability as "development and use of groundwater resources in a manner that can be maintained for an indefinite time without causing unacceptable environmental or socioeconomic consequences"—a definition adopted from the U.S. Geological Survey (Alley et al. 1999). The concept of sustainability as applied to groundwater has evolved considerably from early safe yield concepts toward a more integrated outlook (Alley and Leake 2004).

Groundwater sustainability is not a scientific concept, but rather a perspective that can frame scientific analysis. Ideally, sustainability is a vision that develops from stakeholders about what level of change caused by pumping is acceptable. The concept of sustainability presents a challenge to hydrologists to translate complex and sometimes vague socioeconomic and political questions into technical questions that can be quantified systematically. The sustainability of groundwater resources may be greatly influenced by management practices, such as managed aquifer recharge. This critical topic is covered in a separate chapter of this eBook.

Groundwater developments exist in a continuum (Pierce et al. 2013). At one extreme of the continuum are developments that can be maintained indefinitely. At the other extreme are those that are clearly mining the resource. The discussions about sustainability fall in the intermediate interval of this continuum, where one is trying to safely develop the groundwater resource for long-term use. Sustainability has many facets. This chapter looks at its connection to the evolving concept of groundwater governance and some key aspects of addressing the time response of groundwater systems.

Groundwater Governance

Recent years have seen considerable interest in promoting responsible collective action by the many people and agencies involved in groundwater—including well owners, public agencies, the private sector, environmental groups, and water consumers. These ideas fall under the general umbrella of "groundwater governance" (Foster and Garduño 2013; Megdal et al. 2015; Global Environment Facility et al. 2016). Governance differs from management in that the latter is what agencies do within the governance framework to implement the policies and plans that have been established.

Effective groundwater governance requires collaboration, meaningful stakeholder participation, and community engagement. A widely-shared understanding of groundwater systems and communication to stakeholders about how critical factors affect groundwater sustainability are also key. The greatest shortcoming of groundwater governance has been called "its failure to grasp the central importance of the human dimension . . . and the consequent neglect of stakeholders in governance and management" (Global Environment Facility et al. 2016).

Raising awareness is essential to get political and stakeholder participation, and to achieve a greater sense of urgency to address current problems and long-term risks. Rather than starting from scratch, discussions of groundwater sustainability can often build on existing frameworks, such as river basin commissions (e.g., the Delaware and Susquehanna Rivers).

A recent novel approach toward groundwater sustainability is California's Sustainable Groundwater Management Act (SGMA). Groundwater governance issues play a very large role in SGMA. For each basin defined as medium or high priority by the state, the act requires new local agencies to self-organize as groundwater sustainability agencies and develop plans to bring the basin into sustainability by about 2040. SGMA defines sustainable groundwater management as a basin operated in such a way so as not to cause "undesirable results," such as chronic depletion of groundwater, seawater intrusion, or land subsidence. Kiparsky et al. (2017) describe some of the institutional challenges.

Based on a review of nine case studies in six states, Babbitt et al. (2018) emphasize the importance of building trust, having sufficient data, using a portfolio of management approaches, assuring performance, and access to funding. After reviewing numerous examples internationally, Alley and Alley (2017) identify 13 factors contributing to good groundwater governance (see Table 1). Among these, they emphasize that the primary solutions are found at the aquifer, watershed, or local level. There's virtually no possibility of getting entrenched groundwater users on board, if they aren't actively involved in the decision-making process. At the same time, an external force is often required to achieve necessary changes and accountability.

Sustainability, Governance, and Time

Managing groundwater resources sustainably requires considering the timescales of the consequences. Society is poorly adapted to balancing environmental issues and economic development over intergenerational timescales, yet these are the timescales of many groundwater systems. Gleeson et al. (2012) suggest setting groundwater sustainability goals for many aquifers on a multigenerational time horizon (50 to 100 years), while continuing to acknowledge longer-term impacts. A key benefit of setting longer groundwater policy horizons is the educational value in fostering increased awareness of the long-term effects of pumping.

Among the most challenging aspects are those associated with capture (defined as the decrease in discharge plus the increase in recharge resulting from groundwater withdrawals). Capture is often considered synonymous with (or dominated by) streamflow depletion (Barlow and Leake 2012). It also manifests as reduced groundwater discharge to (or induced infiltration from) lakes, wetlands, and other surface water bodies, as well as reduced transpiration from groundwater.

In some areas, capture can include increased recharge caused by water-table declines in areas

Table 1. Factors contributing to good groundwater governance (from Alley and Alley 2017).

- Recognizing surface water and groundwater as a single resource
- Active engagement of local stakeholders in the decision-making process
- Pressure from external bodies to achieve suitable and workable solutions
- Public education on groundwater and its importance
- An emphasis on public guardianship and collective responsibility
- Consideration of groundwater within other policy areas, such as agriculture, energy, and land use
- Adequate laws and enforcement
- Fully funded and properly staffed groundwater management agencies
- Characterization of major aquifer systems
- Effective and independent monitoring of groundwater status and trends
- Recognizing the long-term response of groundwater systems
- Accounting for interactions between groundwater and climate
- Community leadership

where high water tables previously precluded infiltration. By examining numerous groundwater modeling studies, Konikow and Leake (2014) found that on average about 85 percent of the water pumped in these systems came from capture and 15 percent from storage depletion. The significance of capture relative to storage depletion comes as a surprise to most people.

Groundwater modeling is an essential tool to estimate how capture plays out over time and is also an example of the previous statement by the Global Environment Facility about the neglect of the human dimension in groundwater governance and management. In the past, groundwater models have been developed largely or exclusively by a single group, with limited input from those who have a stake in the outcome. A peer review generally occurs at the end of the model construction process. This has led to models with limited buy-in from stakeholders, undermining their usefulness.

It is increasingly recognized that to build trust in contentious situations, models should be developed through a more collaborative, inclusive, and transparent process with major stakeholder groups more actively involved in groundwater model development. In this way, stakeholders more fully understand the purpose of using a model, the data used in its construction, and model limitations and uncertainties. While the process requires greater commitment of time and resources, the model is much more likely to be trusted by the majority of stakeholders. Recent examples of collaborative modeling include the Upper San Pedro River in Arizona (Richter et al. 2014) and the Wood River in Idaho (Wylie 2017).

Climate is another temporal issue associated with groundwater sustainability, commonly connected with the concept of resilience (Foster and MacDonald 2014). NGWA defines resilience as the capacity of a groundwater (or water-resources) system to withstand either short-term shocks (e.g., drought) or longer-term change (e.g., climate change). When discussing resilience, the timeframe under consideration should be defined. Resilience applies to both water quantity and quality.

Climate variability and change influences groundwater systems both directly through replenishment by recharge and indirectly through changes in groundwater withdrawals. These relations can be complex (Taylor et al. 2013). Groundwater is commonly taken for granted as a buffer storage that can assure water availability during times of drought. The reality can be quite different, however, with groundwater management failing to adequately consider the natural cycles of wet years and dry years, let alone potential long-term climate change. As a result, groundwater may fail to meet its expected role in drought mitigation and droughts simply intensify the overexploitation of groundwater resources.

Key challenges in good groundwater governance are maintaining awareness during wet periods of the importance of groundwater as a backup resource, and working toward laws, regulations, and incentives that encourage use of surface water during wet periods and prepare for increased groundwater use during droughts (Alley 2016).

Final Thoughts

The long timescales and uncertainties of groundwater systems suggest the use of adaptive management approaches. Adaptive management or staged decision-making is commonly presented as an approach to making choices about long-term management under uncertainty. Although the effectiveness of adaptive management for addressing groundwater depletion remains largely untested, and if misapplied may become a rationale for early inaction (Bredehoeft and Alley 2014), setting interim short-term and long-term goals with planned revisits is an obvious need in many situations. With large uncertainties, long timeframes, and numerous stakeholders, a key challenge is to get started as soon as possible.

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Land Subsidence

By Michelle Sneed

Introduction

Land subsidence in the United States is inextricably linked to the development of groundwater—one of the nation's most valuable natural resources. More than 80 percent of the identified subsidence in the United States is a consequence of anthropogenic impact on water resources. Three processes account for most of the water-related subsidence—the compaction of aquifer systems, the drainage and subsequent oxidation of organic soils, and the collapse of subsurface cavities (sinkholes).

The compaction of aquifer systems that are, at least in part, composed of unconsolidated fine-grained sediments and have undergone extensive groundwater development is the leading cause of subsidence in the United States (Galloway, Jones, and Ingebritsen 1999).

These susceptible aquifer systems deform elastically and/or inelastically as pore spaces expand or contract in response to groundwater-level changes. Seasonally fluctuating groundwater levels can result in a few centimeters of elastic (reversible) land subsidence and uplift. Long-term groundwater-level declines can result in a one-time release of "water of compaction" from the pore spaces of fine-grained sediments. Accompanying this release of water is a predominantly inelastic (permanent) reduction in the pore volume of the compacted fine-grained sediments, and hence an overall reduction of the aquifer-system volume, which is expressed as land subsidence (Galloway, Jones, and Ingebritsen 1999).

This "water of compaction," and the space it once occupied, is not restored should water levels (heads) recover to pre-compaction (preconsolidation) levels. In addition to the loss of water and aquifer-system storage capacity from permanent compaction, differential subsidence can alter land-surface slopes, stream gradients, erosional and depositional patterns and volumes, water depths and temperatures, and can cause riparian corridor and wetland migration toward subsiding areas.

Although these effects to natural systems are of concern, most of the attention regarding subsidence-related damages has focused on engineered structures including aqueducts, levees, dams, roads, bridges, pipelines, and well casings, or hazards associated with flooding and ground failures—surface faulting and earth fissuring. The mitigation of subsidence damages and hazards can be costly and managing groundwater resources in sustainable ways requires avoiding permanent compaction by maintaining heads above preconsolidation levels

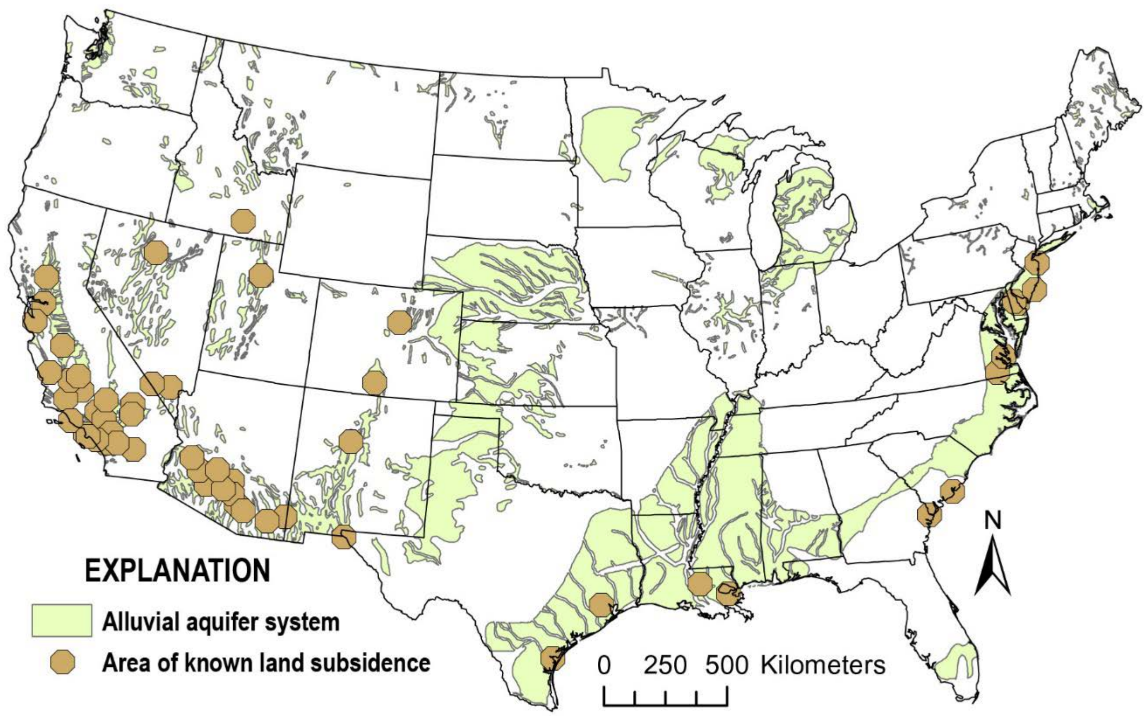
Affected Aquifer Systems

The withdrawal of subsurface fluids from alluvial aquifer systems has permanently lowered the elevation of more than 123,000 km^2 of land and waterways in more than 50 areas in the conterminous United States—an area larger than Pennsylvania (Figure 1). Not surprisingly, subsidence attributed to aquifer-system compaction in the United States generally is largest in magnitude in the arid and semi-arid West, where surface-water availability is limited, and groundwater is extensively used for irrigating agriculture and to support industries and growing populations.

While most of the subsidence occurs in inland basins in the West, subsidence in coastal basins in California, Texas, Louisiana, and near the Chesapeake Bay plays a role in the relative rise of local mean sea level. In these coastal settings, local mean sea level is affected by land movements attributed to subsidence and sea-level changes attributed to eustasy (Eggleston and Pope 2013).

Each of the affected aquifer systems in the 54 areas shown in Figure 1 is composed of a large thickness of unconsolidated deposits with a substantial aggregate thickness of fine-grained sediments, and most contain a laterally extensive confining unit or units. In the West, groundwater withdrawal for irrigation has been the principal water use. Where supplies were available, surface water has been imported to mitigate effects of groundwater depletion (Galloway and Sneed 2013). The relative importance of historical groundwater uses for industrial, municipal, or irrigation applications has varied with land-use changes and population growth.

Figure 1. Fifty-four areas of known land subsidence due to subsurface fluid withdrawal in the conterminous United States (modified from Clawges and Price 1999 and Galloway, Jones, and Ingebritsen 1999).



Measurements and Analyses

Land subsidence and aquifer-system compaction measurements have been acquired using various methods. Subsidence is calculated by differencing the repeated elevation measurements derived from spirit-leveling surveys, or the repeated distance measurements between the ground and satellites or aircraft using campaign Global Positioning System (GPS), continuous GPS (CGPS), or Interferometric Synthetic Aperture Radar (InSAR) methods. The only method to directly measure aquifer-system compaction is by the use of a borehole extensometer. Aquifer-system compaction is tracked by repeated distance measurements between the extensometer element anchored at depth, and a reference point on or near the land surface.

Some of the oldest subsidence determinations were made by repeated spirit-leveling surveys, which are used to calculate the relative elevation changes of geodetic monuments, or benchmarks, over time. Spirit-leveling surveys were later followed and largely supplanted by campaign-GPS surveys, which are used to calculate the absolute elevation changes of the benchmarks. The loss in vertical resolution using GPS surveys (typically 20 mm; Zilkoski, D'Onofrio, and Frakes 1997) compared to spirit-leveling surveys (6 mm over a few miles using stringent surveying procedures; National Oceanic and Atmospheric Administration 1984) often is less important than the lower cost of GPS surveys, particularly for larger areas. These spirit-leveling and GPS surveys typically consist of dozens of locations within a monitoring network. The spatial and temporal resolutions of measurements derived from these surveys tend to be fairly low due to the high costs.

The installation of CGPS stations began in the 1990s but blossomed in the 2000s. By 2018, more than 1000 CGPS stations in western North America were operated by various scientific research consortiums or other groups who generally make these data available to the public. Data are continuously collected using antennas that commonly are attached to pipes anchored to bedrock or grouted a few to dozens of meters below the surface and are designed to last for many years. The measurement resolution improves with observation time, although the day-to-day height of some stations can vary by tens of millimeters. The temporal resolution of CGPS methods, generally 15-30 seconds, is the highest of all subsidence monitoring methods, which facilitates detailed time-series analyses, but the limited spatial resolution and relatively short measurement history currently prevent regional and longer-term (decadal) subsidence analyses.

InSAR is a satellite or aircraft-based remote sensing technique that can detect centimeter-level land deformation over large areas at a spatial resolution of 90 m or smaller—a spacing so dense that geological control of subsidence extent is s ometimes revealed (Amelung et al. 1999). Satellite platforms have the advantage of large spatial coverage, but the signal-to-noise ratio can be small, particularly where subsidence magnitudes are small.

Comparatively, aircraft platforms have smaller spatial coverage and a larger signal-to-noise ratio and spatial resolution, due to the use of high-power instruments transmitting from a lower altitude than an Earth orbit trajectory. Satellite platforms are better suited for regional subsidence assessments, whereas airborne platforms are better suited for subsidence assessments along linear infrastructures or other local features. Additionally, satellites tend to orbit the Earth on a fixed schedule, whereas aircraft can be deployed on a user-prescribed schedule.

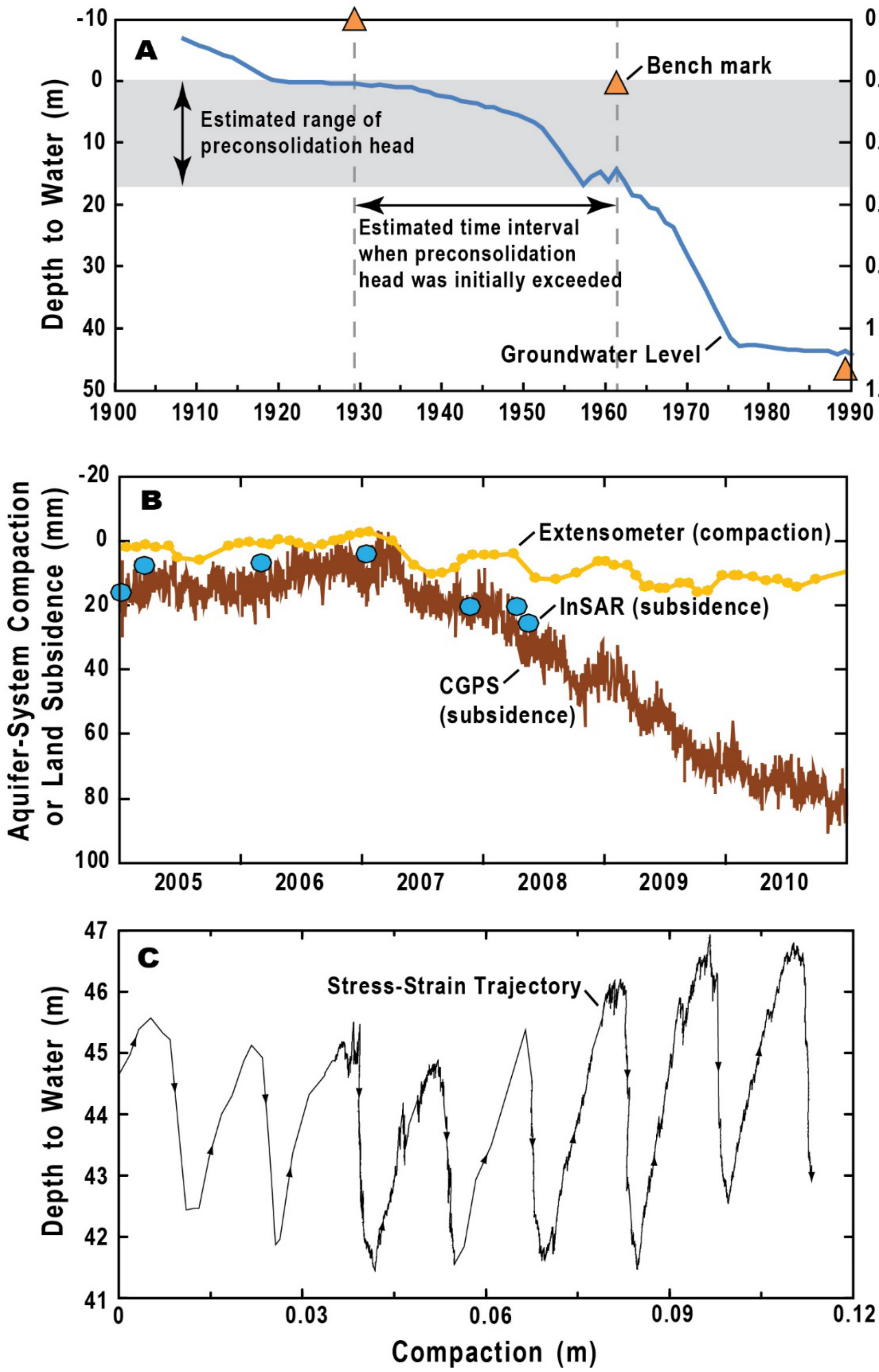
A borehole extensometer is used to directly measure the thickness of a specific stratigraphic interval of an aquifer system. A borehole extensometer is often described as a deep benchmark, and the distance between the deep benchmark (bottom of the extensometer element) and a reference point on or near the earth's surface is tracked. Early designs used a cable as the extensometer element, but were generally supplanted by free-standing pipes, and then counterweighted pipes. The design advances were largely aimed at reducing the friction between the extensometer element and the surrounding casing, which can severely degrade the time-series record.

Measurement resolutions vary, but the most sensitive counterweighted-pipe extensometers are capable of resolving 0.01-0.1 mm (Riley 1969). Multiple position borehole extensometers that incorporate markers anchored to the formation have been used in Taiwan to monitor changes in aquifer-system thickness (Hung et al. 2012). The vertical displacements of the marker positions are tracked using repeated borehole logging on calibrated lines or tapes. This method is capable of monitoring several tens of marker positions in a single borehole at measurement resolutions of about 1-2 mm over depths of several hundred meters. The spatial resolution of borehole extensometers is essentially limited to the installation location and is therefore low, but the temporal resolution can be as high as the logging systems can resolve.

The various aquifer-system compaction and subsidence measurements taken on a myriad of spatial and temporal scales can be integrated to help improve conceptual and numerical models of an aquifer system's response to groundwater-level changes. The earliest survey data provided the basis by which to compare later measurements (of any type) to determine subsidence locations and compute subsidence magnitudes. These early survey data also are critical in determining when subsidence began and—where combined with groundwater-level data—in estimating the level (preconsolidation head) that triggered it (Figure 2A).

The low spatial resolution of those surveys could not capture the subsidence variability of an area that the high spatial resolution of InSAR data can, which can be exploited to better position specialized instrumentation to collect high-frequency measurements of deformation. Extensometers and CGPS stations can be equipped to collect such data for analyses of aquifer-system responses to a range of processes: from daily well operations to seasonal irrigation schedules to longer-term changes in surface-water availability, climate, and land use, among other potentially influential factors. These data from co-located extensometers and CGPS stations can be combined to deduce depth intervals where aquifer-system compaction has occurred (Figure 2B).

Figure 2. Data integration and analyses can help improve conceptual and numerical models of an aquifer system's response to groundwater-level changes. Examples include: (A) Groundwater-level and subsidence data indicate the preconsolidation head was surpassed between 1929 and 1961, corresponding to groundwater levels that ranged from land surface to about 17 m below land surface (modified from Sneed and Galloway 2000). (B) Data from an extensometer anchored 122 m below land surface and from a nearby CGPS station indicate nearly all of the compaction occurred below the anchor depth. CGPS data are used to ground truth InSAR results (modified from Sneed et al. 2013). (C) The stress-strain trajectory shows the absence of expansion during water-level recovery, indicating that residual compaction is an important consideration during the stress-strain analysis (modified from Sneed and Galloway 2000).



The capability to determine the magnitudes of compaction that occur at specific depth intervals is critical for targeting mitigation measures and is important to track as pumping depths and volumes change. These data from co-located extensometers or CGPS stations combined with concurrent groundwater-level measurements can be used for stress-strain analyses (Riley 1969; Sneed and Galloway 2000; Figure 2C) to yield estimates of aquifer-system storage coefficients and preconsolidation head. Experimentation has revealed that the relatively noisy time series of CGPS data render substantially coarser results from stress-strain analyses than do the relatively clean time series derived from extensometers minimally affected by downhole friction.

Finally, integrating measurements provides a measure of ground truth. Even when these data are collected at different locations or times, comparing multiple data sets can give some sense as to measurement quality. Ground truthing InSAR results is particularly important because InSAR measurements are relative and often include substantial artifacts due to atmospheric conditions, anthropogenic activity, or vegetation changes (Figure 2B). Understanding measurement quality is critical for evaluating the utility of the measurements and quantifying the uncertainty of observations and of subsequently developed conceptual and numerical models that use these measurements.

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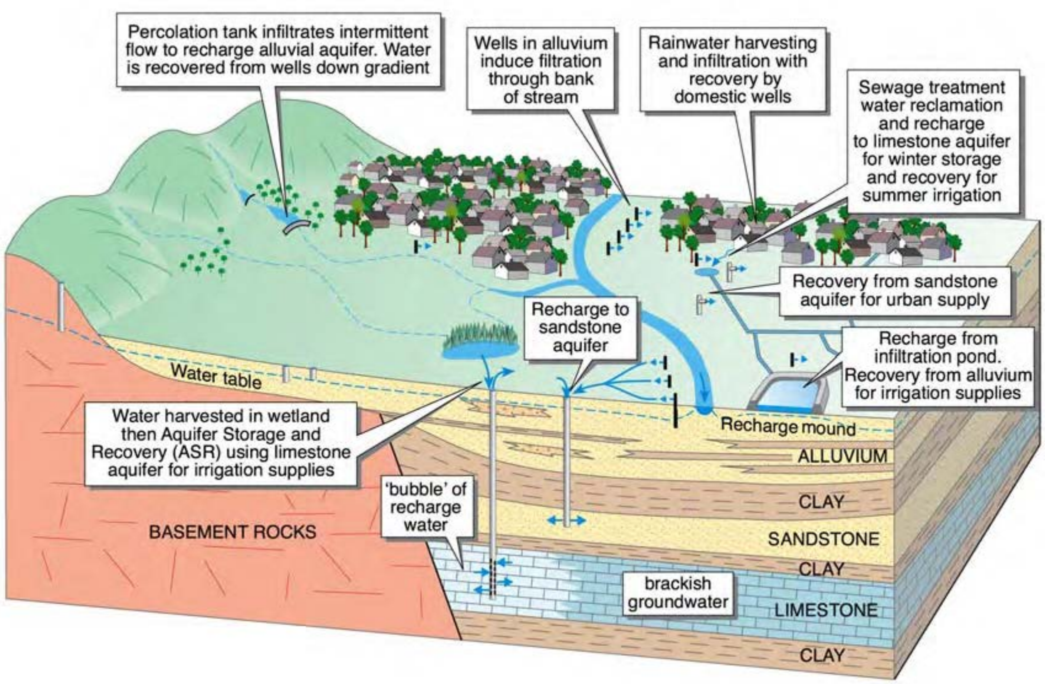
Managed Aquifer Recharge

By Daniel B. Stephens, Amy Ewing, and Stephanie J. Moore

Managed aquifer recharge (MAR), or enhanced groundwater recharge, is the process of capturing water, storing it in an aquifer, and either withdrawing it later during times of water shortage or leaving that groundwater for environmental benefit. MAR encompasses a wide variety of recharge methods and storage management practices (Figure 1).

Surface impoundments and wells are most commonly used to recharge the aquifer. When a well is used to recharge an aquifer and later recover that water from the same well, that process is a type of MAR referred to as aquifer storage and recovery—or ASR. When one well is used for injection but another some distance away is used for recovery, that type of MAR is referred to as aquifer storage, transport, and recovery—or ASTR.

Figure 1. This diagram shows a variety of recharge methods and water sources making use of several different aquifers for storage and treatment with recovery for a variety of uses.



Source: Dillion, et al., 2009 (permission pending)

MAR systems are likely to become increasingly important as growing populations create greater demand for water, especially in urban areas. With suitable hydrogeologic and water supply conditions, MAR can be a practical approach to supplement the portfolio of water managers who deal with meeting peak water demands and drought. Climate scientists predict that in many parts of the country warmer temperatures will further stress our current water delivery systems by creating less natural recharge because of diminished and more intense rainfall, more surface runoff, lower snowpack, and increased evapotranspiration. Where these predictions materialize, MAR will become an even more important water management strategy to consider.

For over a century, MAR projects across the United States have provided numerous benefits, including augmenting water supplies, flood peak mitigation, maintaining in-stream flows, preserving wetlands, improving groundwater quality, and abating seawater intrusion and land subsidence, among others. The key elements of an MAR project include a dependable water source, and a means to deliver the source water to the aquifer. Water sources for MAR projects may include harvested rainwater, surface water, or recycled water.

Harvested rainwater is a viable source for MAR projects, depending on local regulations. Rainwater can be collected from rooftops of individual homes or office buildings and diverted to a dry well, pit, gardens, or greenspace. This source has been used throughout the U.S. and around the world, including many applications in India (Stephens et al. 2012).

Surface water is another source for MAR water. One example is the Central Arizona Project (CAP), which impounds the Colorado River behind the concrete Parker Dam to facilitate diversion to a canal that conveys water for irrigation and domestic uses, including MAR projects in Phoenix and Tucson, Arizona.

Elsewhere, inflatable dams have been used to raise the water level to enhance in-channel recharge and allow streamflow to be diverted into a series of engineered off-channel ponds, quarries, and spreading basins, such as along the Santa Ana River in Orange County, California.

In Albuquerque, New Mexico, an inflatable dam on the Rio Grande diverts surface water to a treatment plant, which then provides water treated to drinking water standards for recharge via an ASR and a vadose zone well (ABCWA 2016).

In Florida, ASR wells inject surplus surface water into the Floridan Aquifer during the wet season and recover it in the dry season (e.g., Reese and Zarikian 2004; Pyne 1995).

Dry wells are essentially wells completed in the vadose or unsaturated zone. These have been used for stormwater management throughout the U.S., especially in extensively hardscaped communities. Dry wells provide benefits not only in the form of stormwater management but also allow for recharge while reducing the potential for groundwater contamination, whereby the unsaturated zone tends to act as a natural filtration system. Now dry wells are being recognized more for their benefit of recharging groundwater, even in the Pacific Northwest. In Phoenix, there are more than 50,000 permitted dry wells, and studies so far have shown no significant deleterious effects on water quality in the alluvial aquifer (Graf 2015).

Recycled water with advanced treatment is another MAR source that is gaining acceptance for potable use. The largest recycled water MAR project in the world is the Orange County Water District's 100 mgd Groundwater Replenishment System, which treats secondary treated wastewater effluent using microfiltration, reverse osmosis, ultraviolet light, and hydrogen peroxide. The advance-treated water is injected into seawater intrusion abatement wells and is also pumped inland to spreading basins. The basin-infiltrated water blends with native water and resides underground for about 12 months before it is recovered by high capacity municipal production wells.

Scottsdale, Arizona, operates a 20 mgd MAR system with advanced treated recycled water, as well as CAP water, which is injected into vadose zone wells to replenish the groundwater which is used as a supplement to surface water deliveries (Gastelum et al. undated). ASR wells are also an important part of Scottsdale's MAR system to recharge recycled water and CAP water and withdraw that water for later use (City of Scottsdale 2018).

Secondary treated wastewater effluent is also used for MAR, as this water is discharged from treatment facilities to stream channels, wetlands, and ponds. The means to improve the quality of the treated water are the natural physical, chemical, and biological purification processes which act on the infiltrated water as it flows through the soil and aquifer before extraction by a well. This has been referred to as soil aquifer treatment (SAT). Most often, MAR with SAT is for non-potable uses such as irrigation, unless the water extracted from the aquifer receives sufficient advanced above-ground treatment. Two good examples include the Donald C. Tillman Plant with Tujunga Well Field near Los Angeles (ASCE 2001) and the Sweetwater Recharge Facility near Tucson (Kimiec and Thomure undated).

The dependability of the water source requires careful consideration in designing an MAR project. Harvested rainwater is highly variable in amount, duration, and occurrence. And, in some locations total annual rainfall is likely to decrease over the next few decades. Likewise, stormwater runoff may diminish and become more variable in the future. Consequently, along the fronts of high mountains more runoff will be lost unless it is captured and stored via MAR projects. Surface water diversions from rivers and streams for MAR projects will become less reliable as an MAR source where climate change lowers streamflows. If streamflow volumes decline appreciably, uses of water other than for MAR may have higher priority. Computer models show future flows in the Colorado River could decline by less than 10% to as much as 45% (Vano 2014), almost certainly reducing water available for MAR and making it difficult for managers to plan for future water availability.

Recycled water is one of the most dependable sources for MAR projects. Municipal potable water production in the future will likely remain stable or increase with population growth, leading to a relatively uniform outflow from water treatment plants that could be used for MAR. However, the quantity of wastewater available for MAR could be threatened if water conservation leads to significant declines in municipal water use.

After an adequate water source is identified which meets the projected demands, then next steps may include conducting investigations to identify potential sites suitable to receive, store, and recover the water. Suitable sites typically have an aquifer with low salinity so that the recharged water does not become affected when mixed with the native water, at least moderate permeability so that the recharge water can be recovered, and in unconfined aquifers a depth to water which allows for adequate storage.

Other important steps include evaluating alternative project designs to deliver the water to the aquifer and transport it to where it is needed, acquiring the necessary land, addressing regulatory and legal issues such as water rights, predicting chemical composition of the produced water, and evaluating project costs. Costs need to consider infrastructure, operations and maintenance, land acquisition, type of storage, water treatment, and transmission.

One of the more common problems in operating MAR projects is clogging of the water delivery systems. Water retention basins often become clogged over time due to the accumulation of silt/clay layers on the bottom of the basin, entrapped air in the pore space, biological activity, and chemical precipitation. This clogging layer usually needs to be removed periodically by draining the basin for sediment excavation or by an underwater unmanned vehicle. Vadose zone wells, dry wells, and ASR wells also are affected by clogging and often need to be redeveloped to maintain sufficient infiltration rates.

If recycled water is selected as the source for MAR, educating the public on the project and seeking comment can be a critical element of an MAR project that should be addressed in the earliest stages of the project (e.g., Hartley 2006).

While MAR projects in general seem to be rather straightforward, there have been unintended consequences necessitating careful attention to impact assessments. For example, at one site in Arizona where nitrate naturally accumulated in the soil, ponded MAR water flushed these salts from the soil, causing nitrate concentrations in the aquifer to increase far above those in the infiltrated water or in the aquifer initially. In Tucson, when the Colorado River (CAP) water arrived and was placed into the existing aged water distribution system which had been conveying local groundwater for decades, the tap water became highly turbid. Geochemical interactions between the recharged water and ambient groundwater have led to unacceptable levels of arsenic and uranium for ASR wells in South Florida (Arthur et al. 2002). And in California there is a policy against anti-degradation of existing groundwater, so that using even potable water for recharge which has any foreign constituents could be precluded by state agencies.

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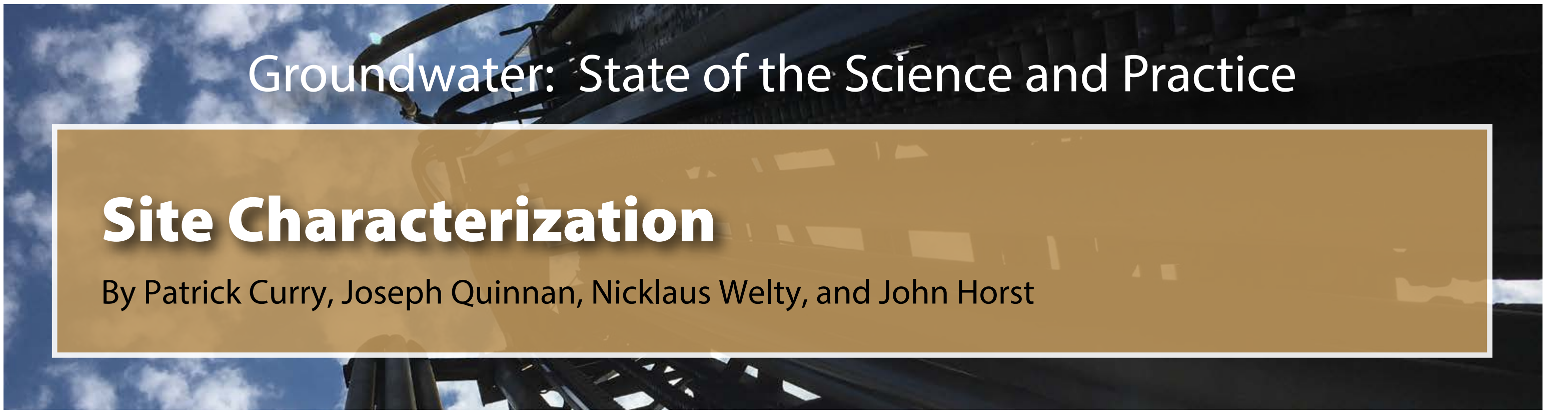


Site Characterization

By Patrick Curry, Joseph Quinnan, Nicklaus Welty, and John Horst

Introduction

In the 1980s and 1990s—the early days of site characterization—the remediation industry leaned heavily on lessons learned from the water supply industry. Groundwater impacts were evaluated with monitoring wells; source areas were characterized with a handful of surface and vadose zone samples; hydrogeologic characterization consisted of pumping tests, point permeability testing, and groundwater elevation flow maps.

The interpretation of this data relied heavily on large-scale averages and steady state assumptions. The concept of dispersion was incorporated with large-scale groundwater hydraulics to provide simplified predictions of contaminant migration. Together, these "low resolution" approaches worked very well for water supply but because they weren't tied to aquifer structure or contaminant characteristics, they obscured important details that could be critical to the success of a remediation effort.

Beginning in the 2000s, the concept of remediation hydraulics began to emerge (Payne et al. 2008) along with a renewed interest in concepts like sequence stratigraphy (Schultz et al. 2017). This began to place more emphasis on how the hydrogeologic structure of an aquifer (aka hydrostratigraphy) controls and focuses contaminant transport.

Today, practitioners are coming to rely on high-resolution site characterization (HRSC) methods. These methods integrate dynamic, real-time, soil, and groundwater sampling with hydrostratigraphic interpretations and permeability mapping in three dimensions. Such investigative tools provide focused windows into contaminant plumes in groundwater, allow mapping of mass transport zones and mass storage zones, and enable implementation of the right technology based on a flux-based perspective to achieve better remedy performance at less cost (Horst et al. 2017).

The depositional patterns that create an aquifer create discrete transport zones where groundwater velocities measure in the hundreds of feet per year. Equally important are the lower permeability zones that support either slow advection or storage, where diffusion dominates, and contaminant mass can sit on the sidelines with velocities measured only in feet per year (Payne et al. 2008; Suthersan et al. 2016).

Experience at sites where we have developed a flux-based conceptual site model (CSM) show that greater than 90% of the contaminant mass flux often occurs in transport zones, whereas the silt and fine sand making up slow advection zones accounts for 9% or less of flux, and the clayey storage zones make up less than 1% of mass flux at any given site (Arcadis 2017).

This "three compartment model" (Horst et al. 2017), involving the identification of storage, slow advection, and rapid transport hydrofacies, can now be mapped at a site in near real-time and at high resolution. When co-located with contaminant concentrations, the outcome is a quantitative, contaminant flux-based CSM that often shows the majority of mass transport (80-90%) occurs within 10% or less of the plume cross-sectional area (Suthersan et al. 2014; Guilbeault et al. 2005). This is a quantum leap forward allowing for simple graphical representation of mass flux that considers all compartments but helps focus remedies on the mass that drives risk at a site.

One question that frequently comes up is the cost of HRSC methods relative to conventional approaches. The advent of new "real-time" characterization technologies has removed critical barriers of cost and lengthy turnaround time that were the hallmarks of conventional characterization. Mobile labs that operate on a daily rate allow for a high frequency and adaptive sampling approach where the emphasis is on optimizing the number of borings, not limiting the number of samples collected.

The introduction of multiple real-time direct push tools in the 2000s have all added to the HRSC practitioner's toolbox. The ability to see data as it is captured in real-time has shifted the paradigm to be both efficient and high resolution. If you can use direct push drilling methods at a site, the possibilities are seemingly endless. More difficult drilling conditions (e.g., bedrock, very deep, etc.) require creative or hybrid solutions, but the approach remains the same—map the distribution of the mass vs. transport and you can develop a mass flux-based CSM.

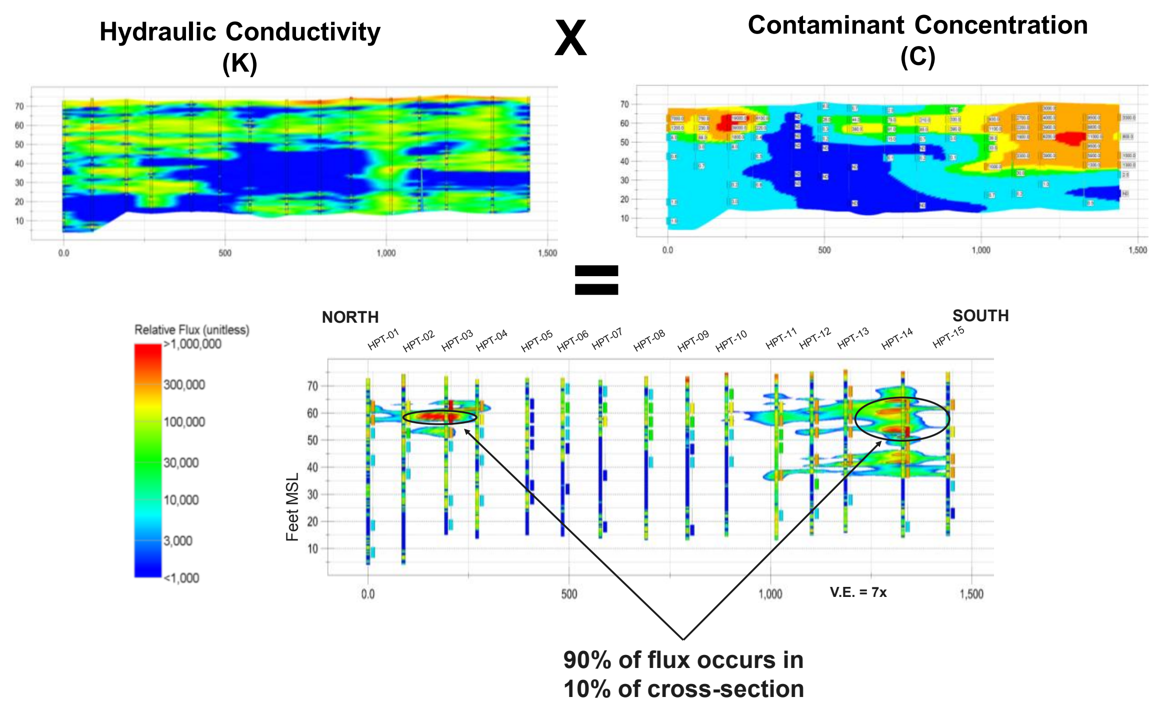
Flux-Focused CSMs

Mass flux puts groundwater contaminants into the proper context; it distinguishes the contaminant mass that is mobile in transport zones from the contaminant mass that is captive within the slow advection and storage zones of an aquifer. The distribution of contamination among the transport, slow advection, and storage zones in an aquifer is indicative of plume maturity and essential to understanding the level of effort and most appropriate strategy required to clean up an aquifer (see also Sale et al. 2013).

Because diffusion is time-dependent, soils near release locations often contain significant mass in the slow advection and storage zones, whereas at the leading edge of the plume, the majority of the mass will be in the transport zones. How the contaminants are distributed through all hydrofacies, both in the source zone and downgradient, are key to defining an effective remedy.

Mass flux is described by the product of co-located hydraulic conductivity, hydraulic gradient, and contaminant concentration (mass/time/unit area). A relative measure of mass flux is provided by simply multiplying permeability by concentration with enough data to adequately characterize each hydrofacies. Tools are available to gain a high-resolution permeability profile and target horizons for groundwater sampling that are biased to different transport zones. The product of this data produces a relative measure of mass flux (ignoring gradient). At more challenging sites, where direct push is not an option, permeability can be evaluated with high frequency sieve analysis or point permeability testing.

Figure 1. Example of a stratigraphic flux transect completed using HPT to measure permeability and vertical aquifer profiling groundwater samples for concentration. Greater than 90% of the mass flux is focused within 10% of the aquifer. This result allowed more accurate focusing of the remediation response.



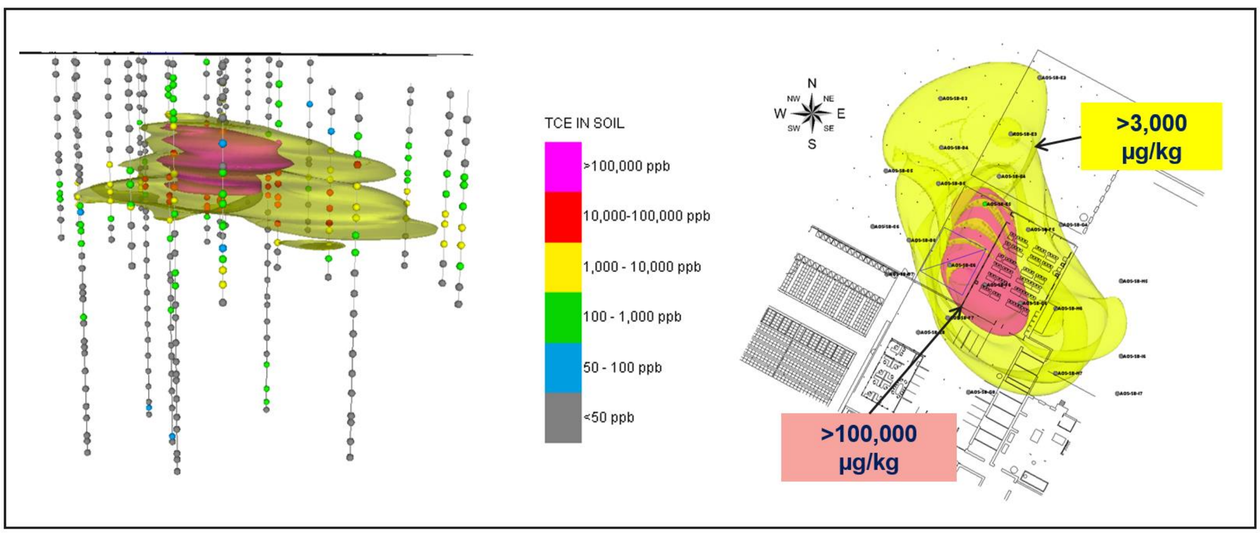
Adaptive Approach to Site Characterization

Historically, site characterization was a lengthy and inefficient process. It seemed that many sites would get locked into a cycle of data collection, evaluation, and reporting, only to identify data gaps that necessitate another round of investigation. This cycle could run on for years, at high cost and exasperation to everyone involved.

Today, with new mobile laboratories capable of running 30 or more samples per day, investigations can be undertaken adaptively using a grid of borings across known source areas or areas of concern—using a first pass to coarsely define the area of investigation followed by step-in and step-out borings based on the real-time laboratory results. Although groundwater samples can be collected from transport zones, recent studies (Curry et al. 2016) have found it more efficient to complete high-resolution soil sampling, through both vadose and saturated sections of an affected aquifer. This has the added benefit of providing data in hard to sample slow advection and storage zones.

Site investigation is typically completed to define risk and/or reduce the footprint and define the "strike zone" for a remedy strategy. The "whole core soil sampling" approach provides a quantitative indication of mass distribution across all hydrofacies. Further, mobile laboratories typically operate on a per-day cost; therefore, the use of a mobile laboratory encourages frequent sample collection, up to lab capacity, without an increase in analytical cost. Thus, mobile labs encourage sufficient sampling density to resolve concentration and mass distribution through hydrofacies, at margins between zones, and through interbedded zones where distribution can be complex.

Figure 2. Example of high resolution TCE source characterization completed with an adaptive grid and mobile on-site laboratory. The highest concentrations were focused within a relatively small area beneath the building tied up in low-permeability clay.



Saturated soil sampling results can be evaluated as a simple screening measure of total contaminant concentration or converted to an equivalent ground-water concentration using the soil-to-groundwater partitioning relationship (USEPA 1996). By calculating the groundwater concentration, based on chemical partitioning, a comparison of source strength to downgradient concentration can be completed to help predict remedy outcome or help to determine if a source remedy is even appropriate for a given site.

The adaptive approach is scalable and can be applied to a small site with a single source area/single plume, or at an enormous site with many areas of concern requiring multiple drilling rigs and weeks or months to complete. The data is communicated to the office daily and allows the near real-time decision making that drives completion of investigation objectives, typically within a single mobilization. Not only does this approach provide a high-resolution flux-based conceptual site model, but by reducing the mobilization/reporting cycles, the overall investigation costs are reduced, and the remedy more focused with a smaller footprint and a reduced operating cost.

The Digital Revolution in Site Characterization

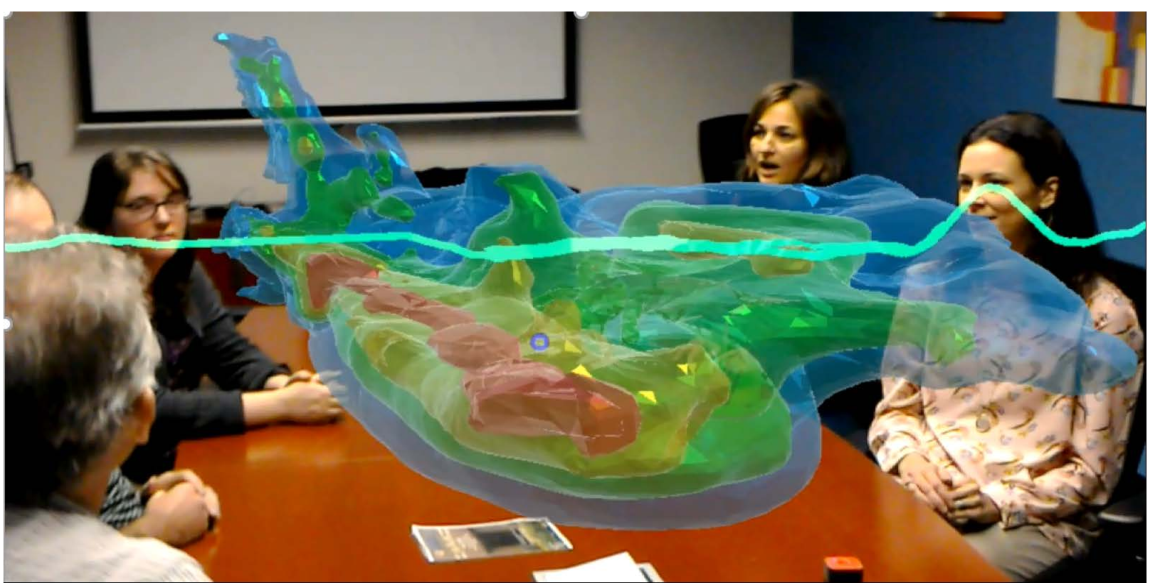
Today, high-resolution site characterization methods can collect more data from a single boring than was collected at an entire site's historical record. This new era of "digital site characterization" results in new ways of working, managing data, developing conceptual site models, communicating, and interacting with stakeholders.

The most fundamental task in site characterization is recording data—a task that now includes options such as tablets, in-well sensors, and unmanned aerial vehicles. These tools collect more data, collect data faster, in some cases collect data autonomously, and deliver higher-quality data than was previously possible. Tablet-based data collection can include boring logs, groundwater sampling logs, field notes, photographs, etc.—all of which can be compiled in a digital format and uploaded to the project team daily. This has the added benefit of reducing transcription time, and in the case of boring logs can be imported directly into 3D models alongside other digital logs from direct sensing tools used at the site.

Our ability to construct and effectively communicate CSMs are being improved with digital technologies. 3D visualization programs are effective at compiling and displaying large amounts of data, but care must be taken so the model does not dictate the interpretation; rather, the geologist should force the model to interpret the data as a geologist would, using calibration and objective quality control.

The next development in 3D models is to move the model off the computer screen and display it as a hologram using augmented reality (AR). Augmented reality combines a live view of the physical world with digital information. Rather than viewing a 3D model on a computer screen, the user is able to interact with the information in a holographic image that appears as a physical object, changing the field of view with the wave of a hand or selecting data behind the interpretation with a voice command or hand gesture.

Figure 3. Example of an interactive holographic conceptual site model depiction using augmented reality (AR) from the perspective of the headset wearer, which can be broadcast simultaneously to a screen or other headsets.



Digital innovations will also improve the social environment of the stakeholders connected to a project by increasing accountability and transparency. One new capability is the evolution of the written report into something more intuitive and interactive. For example, a routine report could be replaced with a live dashboard of site conditions. This dashboard can show the relevant information from the report in an interactive way—like a single map that can be clicked on to change what sampling date or compound is shown, and hydrographs/concentration plots that dynamically update based on the user input. In this way, the report has been turned into a digital conceptual site model. This digital CSM dashboard can be made available to regulatory bodies as well as the site owner, increasing transparency and trust among the stakeholders.

The ability to synthesize vast amounts of data and distill the information down to a useable, easy to understand 3D CSM is the hallmark of the current state of site investigation. Going forward, practitioners must be able to adjust to a constantly shifting regulatory climate (e.g., vapor intrusion), as well as the periodic introduction of emerging contaminants (e.g., PFAS). The constant will be the need for a solid understanding of mass distribution, mass flux, mass discharge, and the geology that controls them. With those pieces in hand, risk can be understood and communicated and, if appropriate, an effective and efficient remedy developed that includes clear objectives and has a better chance of success.

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Surface Water/Groundwater Interaction

By Jeffrey A. Johnson, Ph.D.

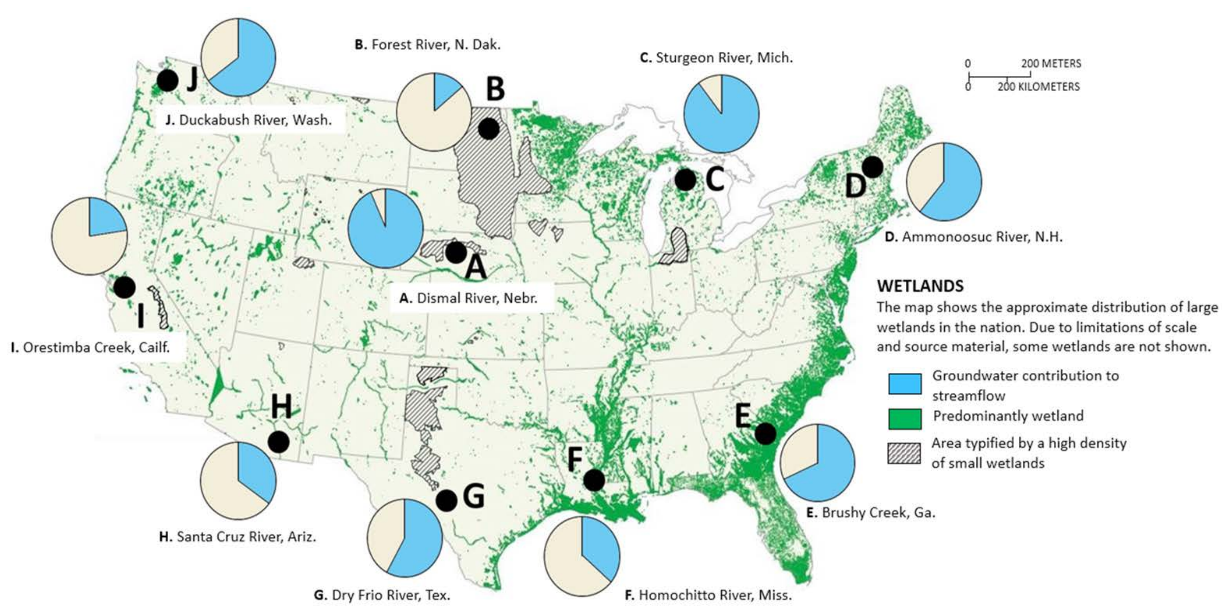
Groundwater and surface water are a continuum within the hydrologic cycle. These different, but linked, hydraulic flow regimes interact to produce physically and chemically dynamic conditions. The nature and extent of these groundwater-surface water interactions vary spatially and temporally.

Common examples of locations where groundwater-surface water interactions play a significant hydraulic role include coastal beaches, shoreline banks, and wetlands. In particular, groundwater-surface water interactions produce conditions that are conducive for an active habitat. Moreover, these areas may also coincide with major urban centers. As a result, contamination has impacted many areas proximal to surface water bodies. Approximately 75 percent of all the Superfund and RCRA sites are located within one-half mile of a surface water body (USEPA 2000). Contaminated groundwater derived from approximately 50 percent of these Superfund sites have impacted the adjacent surface water body (USEPA 2000).

Physically, the interaction of groundwater and surface water occur within a discrete area peripheral to the surface water body, for example along shorelines and the banks and channels of rivers and streams. Estuaries, lagoons, and wetlands cover large areas where groundwater-surface water interactions are critical to maintaining large ecosystems.

It is estimated that groundwater-surface water interactions may influence more than 175,000 square miles of the conterminous United States. Wetlands comprise a significant portion of these areas (Figure 1). In many of these areas, the consistent flux of groundwater to the surface is critical to the sustainability of surface water features. For example, groundwater is the primary source of water for many streams and rivers. As a result, lowering or raising groundwater levels can produce significant effects on the conditions at the surface.

Figure 1. The distribution of wetland areas and the relative contribution of groundwater to select rivers (after Winter et al. 1998).



Where channels convey surface water, groundwater and surface water interact and mix at, and proximal to, the banks and base of the channel. The subsurface zone where these interactions occur is termed the hyporheic zone. This zone is highly dynamic in terms of the movement of water and the chemical and biological interactions that occur. Where surface water levels are not static—such as along tidally influenced coastlines, estuaries, and embayments—groundwater flows in multiple directions in response to the changing level of the water body.

The path of groundwater flow from recharge to discharge and the time that the water is retained below the surface is highly variable. If the water remains at shallow depths, the flowpath to the area of discharge may be only hundreds of feet in length with travel times of months to years. However, if the water flows to deeper depths, the potential for discharge to the surface decreases. As a result, the flowpath to the surface may be more than tens to hundreds of miles with travel times extending over centuries to millennia.

Channels are termed gaining or losing, depending upon the direction of groundwater flow. Gaining channels occur in areas of positive hydraulic gradient and receive groundwater inflow. Losing channels occur where the hydraulic gradient is negative and water flow is out of the channel.

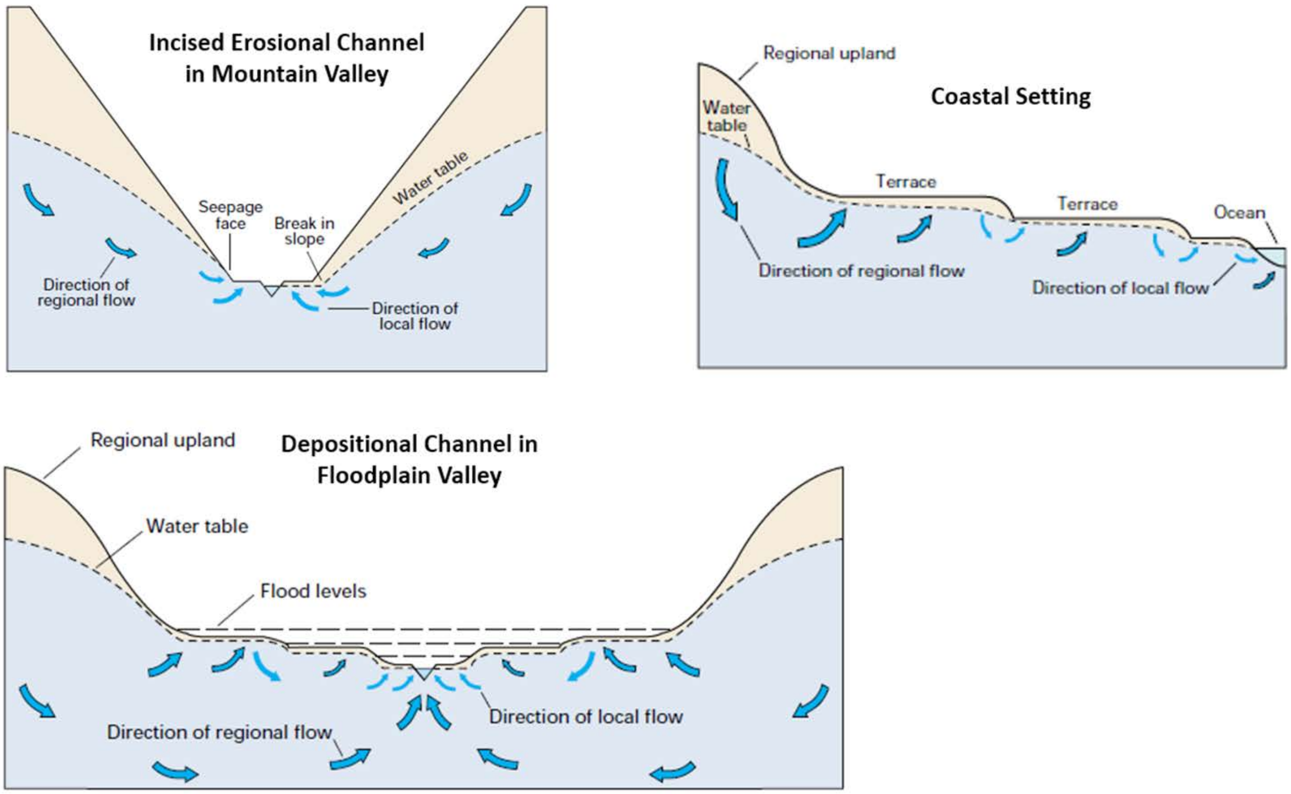
Static water bodies may also act as both discharge and recharge features. The conditions influencing the movement in or out of the basin include precipitation, topography, and the spatial relationship of other hydraulic features. Changing water levels over short periods of time within the surface water body may produce dynamic flow conditions where water constantly moves into or out of the surface water body.

Field studies of the groundwater discharge process in shoreline groundwater zones document that the tide can significantly influence the temporal and spatial patterns of groundwater discharge as well as the chemistry of the near-shore groundwater. In particular, the tidal fluctuations in surface water bodies produce progressive pressure waves in the adjacent groundwater. As these pressure waves propagate inland, groundwater levels and hydraulic gradients continuously fluctuate.

In confined aquifers, the additional weight of the water on the surface in response to the tidal fluctuation increases the pressure on the water at depth. This produces an increase in potentiometric level of the groundwater inland. For this condition, no movement of water between the surface water body and the groundwater may have occurred. However, in unconfined groundwater conditions, water moves directly through the pores network to produce water level fluctuations inland.

The geomorphic setting defines the hydraulic conditions at the groundwater-surface water interface (Figure 2). For example, wetlands form in areas of low relief where the groundwater surface is located near the land surface for an extended period of time. In erosional uplifted settings groundwater discharges into channels forming seeps and springs at breaks in slope, upwelling in terrace deposits, and seepage into the channel. Wetlands are generally not a dominant landform in uplifted erosional settings since the groundwater generally occurs well below the land surface. In particular, the surface water flow in these channels tends to be erratic and "flashy," characterized by periodic elevated flows and stage heights that occur in response to precipitation events. In between these periodic events, the continual discharge of groundwater into the channel produces a base surface water flow in the creeks and streams. With distance, the base flow of the channel increases in volume in response to the accumulating groundwater discharge into the creek and stream. In this setting, the alluvial deposits are relatively thin, coarse in texture, and located proximal to the channel. In many areas, the erosional bedrock surface is in direct contact with the surface water.

Figure 2. Groundwater flow associated with different geomorphic settings (after Winter et al. 1998).



In contrast to uplifted erosional environments, in depositional alluvial settings surface water bodies (channels, ponds, lakes) are encased within a thick unit of Quaternary-age alluvial deposits. Bedrock, if present, generally occurs distal to the channel. Regional groundwater movement is toward the channel, with groundwater moving through the adjacent alluvial deposits and discharging into the surface water body. Within the alluvial deposits water movement may be complex, depending upon height of the water level in the channel. During periods of flooding, water will flow from the channel into the groundwater within the surrounding sediments that form the banks. At lower water levels, which occur over most of the year, the flow will reverse and the groundwater will discharge into the open water channel.

Spatially the areas of groundwater recharge and discharge may become complex, depending upon the channel form and the distribution of the coarser channel deposits relative to finer overbank deposits. In these depositional settings where the ground surface is at or near the groundwater table, large wetland areas may form adjacent to the channel. These areas may become inundated during periods of elevated water, which promotes further groundwater infiltration. Between these periodic elevated water levels, groundwater seepage occurs back into the channels; however, because the hydraulic gradient is typically very low and the sediments are finer in texture, the drainage is slow.

In “coastal” settings along large water bodies, groundwater-surface water interactions can be complex since surface water levels are typically dynamic. Tidal fluctuations are heterogeneously distributed over the globe, varying in magnitude and frequency. For example, in the northwest United States, tidal fluctuations on the order of 10 feet occur twice daily whereas along the Gulf Coast, fluctuations are generally less than 2 feet. Tidal conditions are also associated with large water bodies such as the Great Lakes.

The interaction of groundwater and surface water produce a zone where significant chemical transfer occurs. Processes such as precipitation, sorption, and biodegradation may be highly active in the groundwater-surface water mixing zone due to oxidation-reduction reactions and the dissolution and exsolution of gases. As a result, these zones are areas that naturally filter many dissolved constituents from the water and produce diverse habitats for aquatic fauna.

In particular, the hyporheic zone is an area of complex biogeochemical processes. In this zone, the flow of oxygen-rich surface water mixes with discharging groundwater that is commonly oxygen-poor. As a result, an enhanced zone of biogeochemical activity develops that may impede and transform dissolved metals and organic contaminants.

Measuring the physical and chemical conditions associated with groundwater-surface water interactions is important to evaluating the critical role of these processes to the environment. For example, the flux of water movements can be measured both indirectly and directly (Rosenberry and LaBaugh 2008). These measurements can determine the vertical and horizontal hydraulic gradients. Seepage meters can provide a direct measurement of the water flux within a discrete local area. In addition, conservative tracers can also be utilized to document water movements at and proximal to the groundwater-surface water interface.

With the advent of new technologies to measure water quality continuously, vertical profiling of temperature and other conditions (i.e., salinity) can also be utilized to document the dynamic conditions that characterize groundwater-surface water interactions.

In summary, groundwater-surface water interactions incorporate complex physical, chemical, and biological conditions that occur proximal to the boundary of surface water bodies and groundwater. The conditions active in this zone reflect communication between two relatively independent hydrologic features. These environments are dynamic, changing temporally and spatially. More specifically, groundwater and surface water interactions play an important role in the physical, chemical, and biological processes in lakes, wetlands, and streams. Improved scientific understanding of the interactions between groundwater and surface water bodies is required to ensure the sustainability of these features and their associated habitats into the future.

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Well Construction

By Tom Christopherson

Freshwater is a global resource of immeasurable value. With freshwater making up only 4% of the total water on the planet, and groundwater comprising 90% of that total, preserving the integrity of this resource is paramount to ensure the usability of this resource.

Water well construction techniques have been developed over the years to accommodate the varying aquifer conditions encountered. Groundwater can be found in differing environments and capacities. These may include unconsolidated sedimentary deposits such as sand and gravels and fractured clays or consolidated deposits such as fractured shales, sandstones, limestone, and dolomites. It is even found in igneous rock formations such as fractured granites.

Modern water well construction methods are suited to construct and develop a water source in each of these environments. Direct or reverse rotary drilling with mud, air, or foam is best utilized to remove the cuttings from the unconsolidated and some soft consolidated subsurface material. Other methods such as driving steel casing while drilling with compressed air or a mud-based foam are better suited for hard, consolidated, and even igneous rock formations. For areas dependent on local precipitation stored in saturated or fractured clays close to the surface, boring or augering a large diameter hole to accommodate large diameter well casing may be the best method of well construction. The one thing common to all methods of well construction is utilizing materials and techniques to protect groundwater from contamination.

Many people are familiar with surface contamination from nitrates and agricultural chemicals. Contamination from these chemicals is one of the easiest types of contamination to prevent through proper casing and grouting methods. Other contamination from naturally occurring heavy metals and metalloids can be inhibited through proper construction and development of water wells. When a well is constructed with multiple screened areas across natural boundaries of an aquifer, disturbances of water quality equilibriums occur. Care should be taken not to commingle waters of dissimilar water chemistries to avoid premature failures of the well production or cross-contamination of aquifers.

Proper location is key to adequate well construction. The ideal location is upslope from drainage runoff and at least 100 feet from a source of contamination such as septic laterals, privies, and large quantities of waste (e.g., confined livestock feeding pens). Other things to consider are the porosity of the soils, depth to groundwater, distances from other wells (particularly those known to be contaminated), and accessibility of the site for future maintenance.

Water Quantity and Flow

Water usage needs are another factor in well construction. Determining the requirements for a water well system includes comparing the yield of the well to the water usage and flow requirements of the end users. The well's yield, or amount of water that can be produced and sustained, can be determined through pumping tests.

Usage and flow requirements consist of total daily and peak use demands.

Total Daily Demand

Total daily demand is the total quantity of water required each day. On average, Americans use 60 to 100 gallons of water per day. Domestic daily demand can be estimated by multiplying 100 gallons per day by the number of people expected to reside in the home. More specifically, demand can be estimated by calculating usage needs per fixture. Table 1 provides estimated amounts based on national averages; actual use may vary significantly.

Table 1. Water use estimates for household appliances and fixtures (from *NebGuide* G2149).

Appliance or Fixture	Typical Water Use
Clothes washer — standard	40 to 50 gallons per load
Clothes washer — high efficiency	18 to 28 gallons per load
Dishwasher — standard	7 to 14 gallons per load
Dishwasher — high efficiency	4.5 to 7 gallons per load
Sink faucet — standard	3 to 5 gallons per minute of use
Sink faucet — low flow	2 gallons per minute of use
Toilet — standard	3.5 to 5 gallons per flush
Toilet — low-flush (required Jan. 1, 1994)	1.6 gallons per flush
Shower — standard	6 to 8 gallons per minute of use
Shower — low-flow (required Jan. 1, 1994)	2.5 gallons per minute of use
Garbage disposal	4 gallons per minute of use
Water softener regeneration	50 to 100 gallons per cycle
Backwash filters	100 to 200 gallons per backwash
Reverse osmosis filter	3 to 5 gallons per 1 gallon of treated water

Peak Use Demand

Because water usage needs fluctuate throughout the course of a day, the peak use demand must also be considered. Peak use is the amount required to allow multiple fixtures to operate during a short period—such as showers, laundry, and dishwashing during morning and evening times.

In general, water systems should be capable of meeting peak demand requirements for a period of two hours. A minimum of 10 gallons per minute is recommended for a 2-bedroom, 2-bath home, with an additional 2 gallons per minute added for each additional bed or bathroom.

Should the well itself meet the quantity but not flow requirements, a storage system (such as a tank) may be used to supplement during peak usage requirements.

Table 2 provides estimated flow rate requirements of typical household fixtures. Ideally, the well system should be capable of producing flow rates that exceed minimum flow recommendations to allow for multiple fixtures to be used simultaneously.

Table 2. Typical flow rate requirements for household water-using devices. MWPS (Midwest Plan Service), Iowa State University, Ames, Iowa, *www.mwps.org*. Used with permission: Jones, D. *Private Water Systems Handbook*.

Device	Typical Flow Rate Required for Operation
Automatic washer	5 gpm
Dishwasher	2 gpm
Garbage disposal	3 gpm
Kitchen sink	3 gpm
Shower or tub	5 gpm
Toilet flush	3 gpm
Bathroom sink	2 gpm
Water softener regeneration	5 gpm
Backwash filters	10 gpm
Outside hose faucet	5 gpm
Outdoor lawn sprinkler system	12 gpm
Fire protection	10 gpm — preferred 20 gpm

Well Casing

Sanitary well construction requires watertight casing composed of materials that are compatible with the subsurface and groundwater chemistry of the site.

PVC plastic is a popular option due to its resistance to corrosion and encrustation, its relative ease of cleaning, and the inert properties of plastic on minerals found in groundwater—but can be limited to depths of 500 feet or less depending on the site. PVC casing should not be used above the surface where prolonged exposure to UV light will deteriorate its integrity.

Galvanized and threaded steel casing can be used for wells extending to greater depths to maintain structural integrity—but are not as resistant to corrosion and encrustation and are more difficult to clean than plastic casing.

The attributes of fiberglass casing are a blend of those for PVC well casing and galvanized steel pipe in that fiberglass is more resistant to corrosion and encrustation than steel, has inert properties that don't react to water quality chemistry like PVC, and has good strength and structural integrity like steel pipe.

No matter what material is chosen, the casing needs to be watertight throughout the entire length of the borehole above the intake.

Well Screens

Wells that require a screened opening to retain sand and gravel particles of an aquifer need to be sized accordingly with the proper number of openings to obtain the water volume required while preventing the entrance of sand and gravel into the water column. Screened openings also need to be designed for the ease of development and removal of drilling fluids and debris after the drilling and construction process has been completed. Proper well screen development can reduce or remove contaminants such as coliform bacteria that become trapped in the drilling fluids and are deposited within the borehole.

The amount of screen used is dependent on the expected yield required and the transmissivity of the production zone of the aquifer the well is tapping. Caution should be exercised to avoid "over-screening" a formation. Over-screening occurs when the length of screen footage needlessly exceeds the production zone of a formation and requires extended development of the screened area to remove cuttings and fluids from the borehole.

Screens should be of appropriate material to resist the corrosive and encrustation dynamic that occurs in the screen during pumping of the well.

Drilling and Grouting the Borehole

Quality well construction begins before the drill bit even begins to rotate. If drilling with a mud, the drilling fluid “mud” must contain the proper characteristics to remove the cuttings, stabilize the borehole wall, and retain structural integrity until it is physically removed during the development phase of well construction. The same is true when drilling with foam or compressed air. All of these are considered “drilling fluids” as they remove cuttings, cool the bit, and provide borehole stability.

Once the fluids have been engineered to the proper characteristics, the drilling begins. A sample of the drill cuttings that are carried to the surface by the drilling fluids is collected, identified, and recorded during the excavation phase of the borehole construction. Identifying the cuttings and accurately recording their location in the subsurface is critical for completing the successful placement of annular fill materials and grout seals. Matching the grout materials and designing the length of grout interval depends on knowing exactly where an aquitard begins and ends. It is also critical to know the composition of that aquitard so that the appropriate grout material is matched to the aquitard to provide a successful seal.

Avoid using a grout with a characteristic of shrinking and cracking in an environment that promotes dehydration of the grout column. Bentonite slurry will provide an adequate seal if the water chemistry is not “salty” and is placed below static water level. “Salty” water, with respect to mixing bentonite grout, is water with chlorides over 1500 ppm, or hardness over 500 ppm. Non-slurry bentonite such as chip, chunks, or pellets will provide a seal in the borehole above the static water level, and if they crack they will re-hydrate once water contacts them.

Cement-based grouts provide excellent structural integrity and do not desiccate or crack in the borehole above the static water level, but they do not bond to PVC and can create a micro-annulus between the grout material and casing. They do, however, provide adequate grout seals in “salty” conditions and bond well to steel casing. Cement-based grouts can be engineered to reduce shrinkage and cracking by reducing the water content of the slurry and using additives to retard the heat of hydration. Due to the specific weight of the cement column, caution needs to be taken when grouting PVC casing with cementitious grouts. Adjustments can be made by using a heavier walled PVC casing and installing the grout column in lifts instead of continually pumping grout.

Testing and Maintenance

Once the well is completed and fully developed, a baseline water sample should be taken. The parameters for testing should include analysis for “acute” contaminants such as nitrates, coliform, and E. coli bacteria. These are contaminants that will adversely affect health in the immediate future. In addition, a water chemistry profile should be built by testing the water for pH, calcium, iron, manganese, total dissolved solids (TDS), salinity, magnesium, sulfides, total hardness, conductivity, fluoride, and any other contaminants of local concern such as arsenic, selenium, and uranium. By doing this when the well is new, there is an established baseline that can be used as a yardstick in the future if the water quality changes.

Once a well has been put in use, it is recommended to test for coliform and any contaminants of local concern on an annual basis and anytime there is a perceived change in water quality or the well has been opened. Nitrates should be checked twice a year until there is no change from one testing event to another, and then repeated as needed.

All well construction records should be treated like personal medical records and kept in a safe, accessible place. Records should be updated when changes occur and well maintenance records should include a well log from the contractor who constructed the well detailing all relevant information such as legal location of the well, total depth, static water level, pumping level and at what yield (GPM), casing materials, length and diameter of casing, screen materials, length and diameter of screen, development method and time spent in well development, borehole diameter, intervals of grout, grouting materials, geological log of the subsurface, and date of completion. Pump information such as manufacturer and model should be recorded, and warranty information submitted to the manufacturer as needed.

Records and files should include a list of all repairs and maintenance performed on the water system and by whom. Copies of all water quality sample results and by whom they were analyzed should be retained as well.

Good groundwater quality is a treasure to be valued and promoted. While water cannot be destroyed, the usefulness of water can be wasted if not valued and preserved.

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Well Design

By Marvin F. Glotfelty, RG

The design of water wells may incorporate many different materials and design aspects, but none of those design elements will be appropriate for all situations. As the design of a well is being developed, the well designer should consider the unique circumstances that relate to the particular purpose of the well being contemplated. The design requirements for each well are unique, and attempts to standardize well design with a "cookbook" approach may degrade the overall integrity and value of the well. A universal truth about water well design is that the simplest design for the intended purpose will generally be the best design for that particular well. All the tricks of our trade should be applied when appropriate, but a "keep it simple" policy is the best approach.

Well Design Sequence

At the beginning of the well design process, the well designer may be tempted to embark on a design process that mirrors the well construction sequence—first specifying the surface casing, followed by the borehole depth and diameter, well casing and screen attributes, and annular fill and seal materials. Although it may seem logical to replicate the construction sequence during well design, the best approach is to specify each of the well's attributes from the *inside-out*. That is to say, if an imaginary line were drawn vertically down the center of the well casing and screen as shown in Figure 1, the order of design should start from that imaginary centerline and proceed horizontally outward.

The inside-out design of a well begins with consideration of the well's intended purpose. For example, a well for a single household would have a very different design than a municipal water well intended to produce a much higher flow rate (requiring a larger pump that, in turn, would necessitate a larger casing diameter). A public supply well would also need to meet regulatory standards for drinking water systems whereas an industrial, stock, or agricultural well would not have those requirements.

No matter what the intended purpose of a well (e.g., industrial supply, dewatering, fluid injection, monitoring, or aquifer storage and recovery), its design attributes should address that specific well function. The possible situations and site-specific conditions related to a water well's use are essentially endless, so the design of any particular well should be consistent with the specific requirements for the intended purpose.

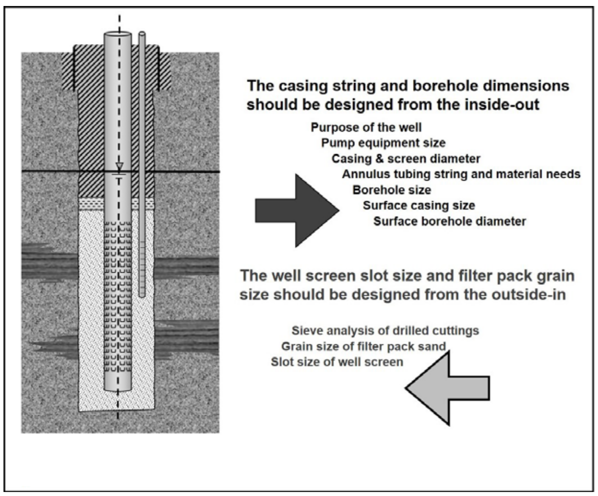
Determination of the specific purpose of the well will govern the necessary pump equipment needed for that well use. The pump type (e.g., submersible pump, line shaft vertical turbine pump) and diameter will delineate the minimum inside diameter (ID) of the well casing that will reasonably contain the pump equipment. For any given casing ID, the outside diameter (OD) will be a function of the casing's wall thickness and connection types (e.g., butt welded, threaded-and-coupled, flush-threaded). The OD of the well casing and the width of the annulus will, in turn, determine the necessary borehole diameter that can accommodate the well (Figure 1).

The result of this domino effect of size constraints, extending outward from the well's centerline, is the surface casing dimensions become one of the last items to be determined during well design, even though it will be the first thing installed during well construction.

The domino effect is reversed for well design influences in the well's screened interval. Adjacent to the well screen, the well design should support robust and efficient groundwater production with little or no sand invasion. This objective leads us to an *outside-in* approach to well design in the screened interval of the well.

As the well's borehole is drilled, cuttings are typically collected at 5-foot to 10-foot intervals. The grain size distribution at selected depth intervals of the borehole can be characterized by sieve analysis of drilled cuttings, and the sieve analysis enables the well designer to determine the appropriate filter pack grain size. Just as the filter pack design is based on the formation characteristics, the well screen slot size is based on the filter pack grain size, so that excessive quantities of filter pack media will not pass through the screen into the well during construction or development. This outside-in approach to the design of a well's screened interval (Figure 1) provides a well design that will be consistent with the local aquifer characteristics, and optimize the well performance.

Figure 1. The philosophy for well design starts at the centerline of the well (dashed line) and proceeds outward.

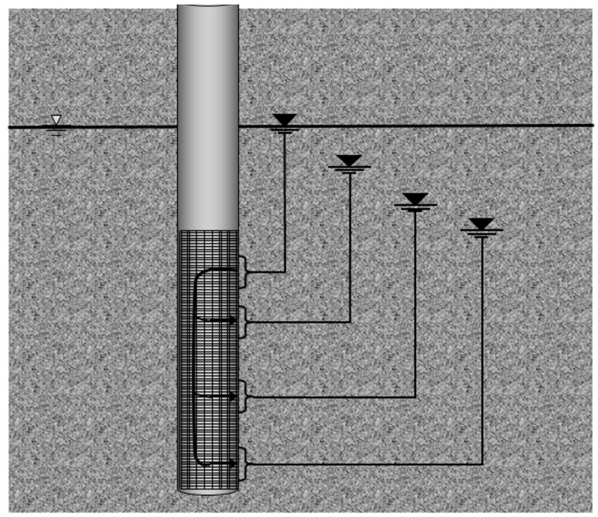


Consideration of Vertical Hydraulic Gradients

In most alluvial aquifers, there is preferential groundwater flow in the horizontal direction, as compared with the vertical flow component. Horizontal hydraulic conductivity is often 10 times to 1000 times greater than the vertical hydraulic conductivity at the same location. This results from the aquifer *stratigraphy* (preferential flow parallel to the layering of sedimentary strata) and *anisotropy* (preferential flow between elongate sediment grains that were deposited in an imbricate pattern, like the shingles on a roof). The stratigraphy and anisotropy can lead to varied hydraulic heads (water levels) in discrete depth intervals of the aquifer.

The varying hydraulic pressure heads within an aquifer can be measured during depth-specific groundwater sampling of isolated intervals in an open borehole, and we sometimes note the heads in the deeper portions of the borehole are lower than the heads in the shallower portions of the borehole (Figure 2). This phenomena is a downward hydraulic gradient, which commonly occurs in many areas. Both downward and upward vertical hydraulic gradients (in areas with confined aquifer conditions) exist, and require consideration in water well design.

Figure 2. Impact of a vertical hydraulic gradient under non-pumping conditions.

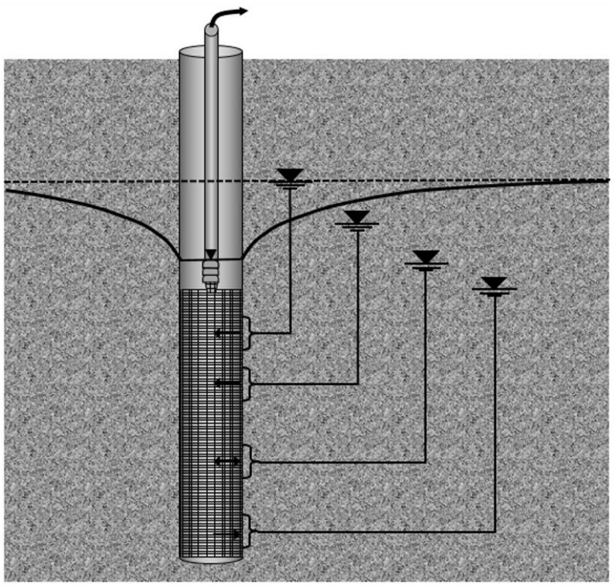


Failure of the well designer to address a vertical hydraulic gradient in the well design will be detrimental to the performance of the well, so this phenomena should get the attention it deserves. In the undisturbed aquifer, even under different hydraulic heads, the stratigraphy and anisotropy of the aquifer will restrict vertical movement of the water. However, after the well has been installed, the different depth intervals will be hydraulically connected to allow water to flow between different depth intervals during non-pumping periods (Figure 2). Mixing of water from different aquifer depths can degrade the pumped water quality, and exacerbate clogging or corrosion problems.

Additionally, the performance of a pumping well (both quality and quantity of water) will be impacted by vertical hydraulic gradients. Groundwater flow from a pumping well (Q) behaves in accordance with Darcy's law ($Q = -KiA$), where the hydraulic gradient (i) is the head difference between the static head in the aquifer and the pumping water level (head) in the well (the change in head per unit of distance). Thus, if we assume equal values of cross-sectional area (*A*) and equal values of hydraulic conductivity (*K*) in all depth intervals of the well, the differences in hydraulic gradient alone will still greatly impact the flow contributions from different depth intervals.

A conceptual pumping well with a vertical hydraulic gradient is shown in Figure 3. Groundwater will be produced from the two upper intervals of the well because the static heads in those portions of the aquifer are higher than the pumping water level within the well. In contrast, no water production at all will occur in the two lower intervals of the well because the static heads in those intervals are at or below the well's pumping water level.

Figure 3. Impact of a vertical hydraulic gradient under pumping conditions.



Well Screen Composition

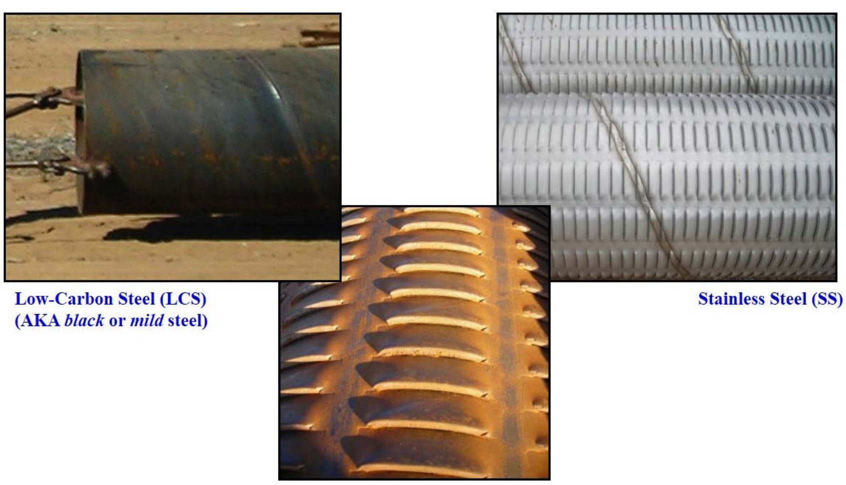
A variety of materials are commonly used for the manufacture of water well casing and screen. The selected material should be consistent with the specific objective of the well design (corrosion resistance, collapse strength, tensile strength, cost, etc.).

Polyvinyl chloride (PVC) is a good material choice for smaller wells because it is relatively inexpensive and essentially inert. However, due to the strength limitations of PVC, steel casing and screen are typically specified for larger and deeper wells.

Common steel types used in water wells include *low-carbon steel* (LCS), *high strength low alloy* (HSLA) steel, and *stainless steel* (SS) (Figure 4). These steel types have different levels of corrosion resistance, different costs, and different susceptibility to growth of biochemical scale or biofilm.

LCS is the most common steel type used for water well construction, as it is less expensive than most other steel types and is a good choice for many well uses. HSLA steel contains a variety of alloyed elements that provide improved corrosion resistance in comparison to LCS. HSLA steel is more expensive than low-carbon steel, but less expensive than stainless steel. HSLA is a good choice for wells where moderate cost is acceptable and a longer-term well life is desired. SS is one of the more expensive steel options, but is a good choice for wells where long-term well life is needed. The higher costs of SS (Type 304L or Type 316L) may be offset by long-term well performance in some cases.

Figure 4. Common steel types used for water well construction.



Low-Carbon Steel (LCS)
(AKA *black* or *mild* steel)

Stainless Steel (SS)

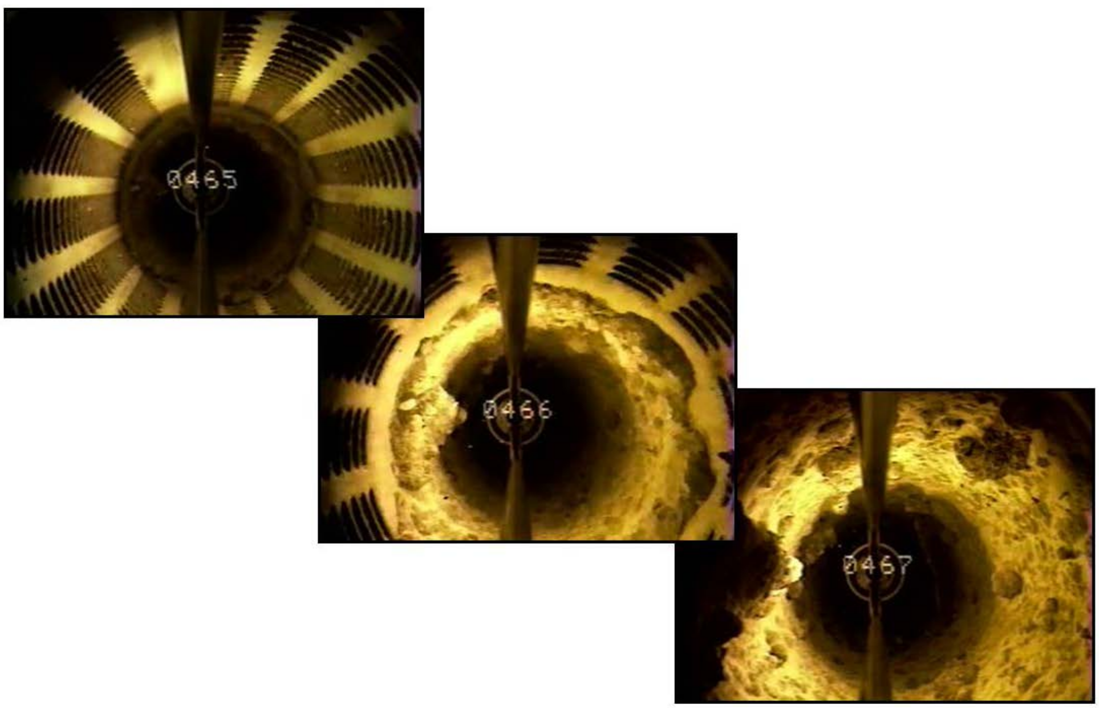
High Strength Low Alloy Steel (HSLA)
(AKA *Corten*, *Kai-Well* or *hard red steel*)

Other steel options, such as galvanized steel and copper-bearing steel, are also commonly used for well construction in some areas.

In addition to having different levels of corrosion resistance, LCS, HSLA steel, and SS well screens will accumulate scale incrustation and/or biofilm at different rates and to extremely different degrees. The naturally-occurring microbial organisms in aquifers seem to favor LCS or HSLA steel over SS, likely due to the differences in metal composition. This has been documented from well videos at many locations, but because each well is in a unique subsurface environment, it is difficult to make a direct comparison between wells at different locations.

However, a well in the El Paso, Texas, area was constructed with alternating steel types (LCS and SS), which allows a direct comparison of scale accumulation between the two steel types. The well has alternating sections of LCS blank casing and SS louvered screen. The intervals of the well with SS louvered screen have minimal scale accumulation, whereas the intervals of LCS blank casing just a few inches away from the SS screen show significant accumulations of scale (Figure 5).

Figure 5. Example well showing different scale growth on surfaces of low-carbon steel (below 466 feet) vs. stainless steel (above 466 feet).



For wells designed to meet critical water resources needs or long-term water demands, the life-cycle performance of a well can offset the cost of more expensive items such as stainless steel screen.

As an example, a 75-year life-cycle economic analysis (M.F. Glotfelty, *Life Cycle Economic Analysis of Water Wells–Considerations for Design and Construction:* Distinguished McEllhiney Lecture for the National Ground Water Association, 2012) was conducted on three hypothetical wells with identical designs—except for the well screen steel type. The averages of a dozen driller bids from recent well installations were used for well construction cost estimates, and actual operational costs from the City of Phoenix, Arizona, were used to provide a representative economic analysis.

The three hypothetical wells were all 1200 feet deep with an 18-inch-diameter well screen composed of either LCS, HSLA, or SS. On the day these hypothetical wells were constructed, the SS well cost estimate exceeded the HSLA well cost by about $210,000 and it was roughly $265,000 more expensive than the LCS well.

Reasonable estimates and assumptions were applied to each well type, based on empirical data from the city's operations staff. The life-cycle economic analysis considered well installation costs; consultant costs; well cleaning costs; operations and maintenance costs for city personnel; pump and motor replacement costs; and electrical costs. When all these costs were rolled up for each well type, it was determined at the end of the 75-year life-cycle period, the least expensive LCS well cost its owner about $3.3 million more than the higher-priced SS well. Similarly, the cumulative life-cycle costs of the HSLA well totaled almost $2.1 million more than the SS well.

Economic life-cycle analyses are applicable only to the unique well use scenario to which they were developed, but this study shows significant cost savings can sometimes be realized from proper (albeit more expensive) initial well design.

Well Maintenance and Rehabilitation

By Stuart A. Smith, MS, CGWP

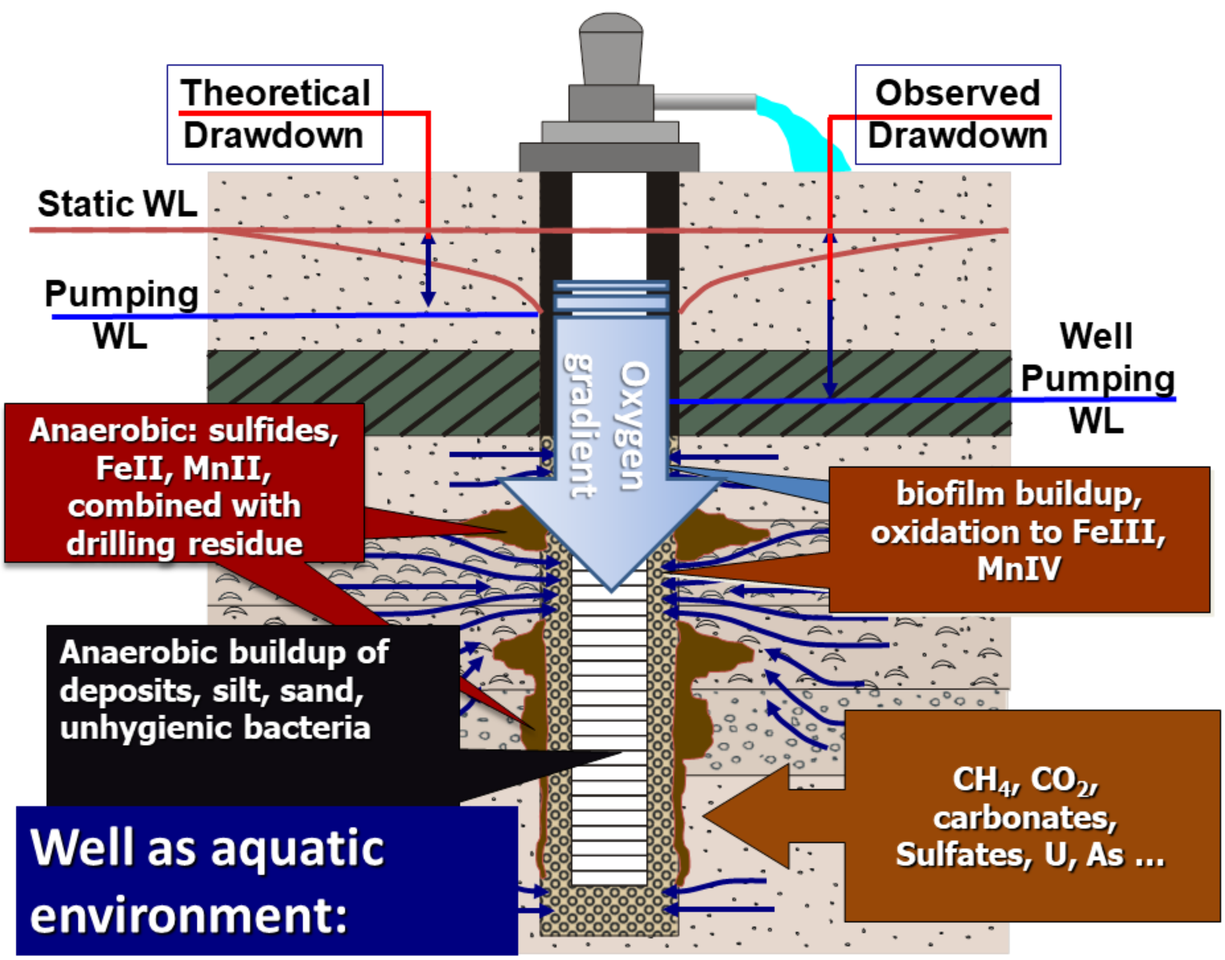
Causes of Well Performance Problems and Failures

Water wells are an interface between the engineered and natural environments (Figure 1), and like other mechanical structures or equipment called upon to operate in the harsh natural environments, are subject to loss of performance due to a variety of natural mechanisms and mechanical wear. An important step in implementing well maintenance is to understand what performance degradation or failure modes are possible. The range of possibilities is large and site-specific.

Prevention of Corrosion, Encrustation and Well Fouling

Well corrosion (including biocorrosion), incrustation, and well fouling (including biofouling) result

Figure 1. The pumping well as ecosystem with typical challenges to performance.



from a complex interaction of the physical-chemical characteristics of pumped groundwater (temperature, pH, redox potential, solids, mineral and metal content), pressure changes in the well, the occurrence of biofouling microflora (virtually ubiquitous), well material composition, and well use.

Prevention or mitigation of well corrosion, incrustation and fouling depend upon:

1. Knowledge of groundwater physical, chemical, and biological characteristics.
2. Preventive design: Reducing intake pressure loss, corrosion potential, exposure to encrusting groundwater if possible, and planning for treatment. Material selection to reduce corrosion is a crucial part of this.
3. Maintenance monitoring for indications of fouling and performance impacts.
4. Preventive treatment in some instances: This is usually conducted when there is a history of performance decline in nearby wells, which appears to be inevitable due to water quality conditions. It is conducted if indicators from maintenance monitoring predict that performance or undesirable water quality impacts will occur.

Preventive Maintenance, Monitoring Methods, and Records

The above-described problems can be prevented and mitigated by effective O&M, but to do so requires valid information on the environment, hydrology, and material performance of the well system produced through a process known as maintenance monitoring. The ideal working methodology is to detect deteriorating effects in time to prevent problems or soon enough to employ the most effective countermeasures. This maintenance monitoring system should include valid information on

- Well construction, dimensions, and hydraulics, including history
- Aquifer environment and hydrology
- Material performance of the well system.

Table 1. Summary of recommended preventive maintenance monitoring parameters.

Hydraulic testing	Flow and drawdown for specific capacity.
	Total amount of pumping time and quantity pumped per year.
	Periodic step-tests for well and pump efficiency linear and nonlinear loss estimates.
	Power and fuel consumption for pump efficiency.
Physicochemical parameters	Total and ferric iron, and total manganese (and other metals as indicated) looking for changes due to deterioration.
	Important cations (Ca, Mg, Na) and anions as identified, including sulfides, sulfates, carbonates, and bicarbonates.
	pH, conductivity, and redox potential (Eh) where possible.
	Turbidity or total suspended solids calculation of product water.
	Calculation of corrosion/encrustation potential—not as a predictor.
Microbial	Total Fe/Mn-related bacteria (IRB), sulfur-reducing bacteria (SRB), slime-forming and other microbial types of maintenance concern as indicated. Also an indicator of total live biomass such as ATP, but not without the others.
Visual/physical	Pump and other equipment inspection for deterioration.
	Borehole TV for casing and screen deterioration.

Information to Be Collected for Preventive Maintenance Monitoring

Table 1 is a summary of useful information to collect about wells for both troubleshooting and predicting problems in preventive maintenance (PM).

In a PM monitoring program (in contrast with a troubleshooting analysis), system water and quality and performance monitoring are compared over time to establish trends. To the information in Table 1 additional information should be added about the wellfield environment useful in interpreting trends (see Table 2).

Table 2 provides a troubleshooting summary guide for well maintenance. Additional information helps in interpreting trends. In preventive maintenance, once a problem such as tendency to biocorrosion is identified, the same choice of parameter monitoring can be employed repeatedly.

To make use of such information over time:

1. A maintenance system must have organized and accessible records.
2. Information collection should start with the project design phase and continue throughout the working life of the extraction and injection system.
3. Records must be regularly reviewed by qualified personnel.

Table 2. Troubleshooting summary guide for well maintenance.

Problem	Review area-regional groundwater conditions	Review design and construction records	Conduct downhole TV inspection	Check SWL and PWL and review histories	Conduct step tests and review history	Test for biofouling parameters and review records	Test for physical-chemical parameters and review records	Test pump mechanical condition and performance	Check for power malfunction
Sand/Silt Pumping		✔	✔	✔	✔				
Silt/Clay Infiltration		✔	✔	✔	✔				
Pumping Water Level Decline	✔	✔	✔	✔	✔	✔	✔	✔	
Lower (or Insufficient) Yield	✔	✔			✔			✔	✔
Complete Loss of Production		✔						✔	✔
Chemical Encrustation	✔	✔				✔	✔		
Biofouling Plugging	✔	✔	✔			✔	✔		
Pump/Well Corrosion	✔	✔	✔		✔	✔	✔	✔	
Well Structural Failure	✔	✔	✔	✔					

In general, maintenance monitoring approaches should be tried and reviewed over a period of time and adjusted, based on experience. They must be implemented as part of a systematic maintenance program involving

- Institutional commitment
- Having a goal of deterioration prevention
- Systematic monitoring as part of site maintenance procedures
- Employing a method of evaluation to determine necessary maintenance actions.

In any case, it has to be recognized that monitoring approaches and responses will be site-specific, and likely will require adjustment during implementation. A minimum of baseline data on each well is needed to assess and interpret its performance through time. Data trends are more reliable if data collection is incorporated into the project plan at the onset.

Records and Software for Preventive Maintenance Monitoring

Records for well maintenance are essential. It is impossible for a manager to effectively remember all data and other information, such as procedures, and personnel turnover requires that records be available if successors are to understand the history of a well or wellfield. Maintenance monitoring assumes that records of data will be kept in order to establish trends. Records may be entirely "analog" hard copy files or combined with a software approach. A variety of options are available. Finally, it is more important to have a deliberate systematic means of collecting, storing, accessing, and assessing data than a sophisticated means.

Maintenance and Rehabilitation Treatment

First, the application distinctions: A maintenance treatment is intended to prevent a large decline in well performance or water quality. A rehabilitation treatment is one performed to reverse a notable decline and structural damage to a well. The difference is a matter of degree, with maintenance treatments being on the whole less intense. Figure 2 summarizes necessary treatment components.

Chemical choices should depend on educated evaluations of effectiveness, cost-effectiveness, and reactivity, not just on vendor promotion. A thorough review of available literature and presentations on the topic will reveal differences in opinion on chemical choices. Future work may change the conclusions of this and other references.

Figure 2. Necessary components of well maintenance and rehabilitation treatments.

1. *Effectiveness:* The chemical solution chosen should be suitable for dispersing the developing clogging materials.
2. *Cost-effectiveness:* Cost is frequently cited as an issue in choices made as to whether to use chemicals and electing which ones and how much to use. A better comparison is cost-effectiveness (which factors in results).
3. *"Do no harm":* Do not aggravate the problem by leaving nutrients such as phosphorus compounds or solid byproducts behind. Evaluate reactivity with groundwater dissolved solids.

Chemical types used can be categorized as acids, sequestrants, and biocides. Typically, no one chemical type will address all incrustation and biofouling removal, suspension, dispersal, and repression needs. The space allotted to this chapter is wholly inadequate to discuss them adequately, and the reader should do further research.

Acids: Typically, harsh mineral acids are avoided in maintenance treatments, using organic blends oriented toward biofouling removal instead. Rehabilitation may also employ these acids, but amended hydrochloric acid may be needed to adequately remove accumulated metal oxide and sulfide solids. Blends of acid types are available or can be made to take advantage of the better properties of each.

Sequestration: In well treatment these compounds are most properly used in low concentrations in chemical blends as aids in acidizing mixtures to retain biofilm and metal oxide components in solution for removal, once they are dissolved and dispersed in the water column. They come in numerous forms. Phosphorus-containing compounds in this category should be strictly avoided. Others should be evaluated based on chemical reactivity and effectiveness.

Biocides: These agents are used in an attempt to reduce microbial populations. Of these, hypochlorite chlorine compounds are most commonly recommended for well preventive maintenance (PM) treatment, although peroxide compounds and chlorine-releasing buffered commercial compounds have some application. In contrast to past reliance on strong solutions, milder (<200 mg/L) hypochlorite solutions, properly mixed to favor the hypochlorous acid ion form in solution, are more effective. Sodium dichloroisocyanurate (NaDCC), in various commercial forms, releases biocidal HOCl without a pH-adjustment step and is usually employed as a finishing step, not alone.

The chemicals mentioned are all reactive to some degree and pose risks to skin, mucous membranes, and other soft tissues of humans, and potentially to the environment if handled improperly. No well maintenance or rehabilitation project should employ personnel or contractors in well cleaning who cannot clearly demonstrate competence in relevant chemical knowledge (including knowledge of mixing and application).

Development in Preventive Maintenance Well Treatment

It should also be emphasized that all chemical mixtures are far more effective with adequate mechanical mixing and development, and should be specified based on an adequate analysis of the problem. Well development, which is the mechanical agitation of fluids in a well that is intended to improve hydraulic conductivity, is as crucial in maintenance treatment as in well rehabilitation, but well redevelopment in rehabilitation requires a combination of longer and higher-energy redevelopment.

There are numerous types of well redevelopment methods, most the same as in initial well development, but some specialized to rehabilitation and intended to apply more concentrated energy.

In some cases, redevelopment may be sufficient on its own, or the only treatment permitted. Where possible, lighter chemical treatments augmented by more effective redevelopment are preferred over highly concentrated chemical doses and insufficient development.

Schedule of Well Maintenance Activities

A maintenance monitoring schedule should be based on the principle of establishing a data baseline and then settling into less-frequent (or more intense) PM activity if conditions warrant. Various recommendations exist (again, research beyond this brief chapter). Likewise, schedules of PM treatments should be established based on site-specific knowledge of clogging conditions.

Some wells and wellfields are relatively trouble-free or so well known that a few parameters need to be monitored (flow and drawdown are absolute minimum choices to establish specific capacity). At the other end of the continuum are wells subject to intense chemical and biological attack, such as those involved in management of groundwater contamination. These may require either intense monitoring or just frequent treatment.

Economics of Well Maintenance

The primary obstacle to initiating a robust program of well maintenance is the perceived cost involved in doing so. Going from a regime of little or no maintenance action (also known as "neglect") does involve some cost.

O&M management benefits from taking a "long-view" approach to O&M cost-effectiveness calculations, i.e., to consider cost-effectiveness on a life cycle cost basis.

In developing or redeveloping economies, replacement of assets once installed may be very difficult. In this case, maintenance is mandatory if the well, for example, is to continue functioning.

Generally, it is demonstrated that, in both life cycle and annual variability, life-cycle costs are lower for utilities employing routine maintenance practices. Regardless of the cost differential, the predictability of maintenance costs has many advantages when compared to reacting to well deterioration in a crisis-management manner.

The Way Forward from Here

Field and laboratory research relevant to well maintenance has been ongoing in a rather accelerated way since about 1980, with many technical advances, and asset management principles added in the last decade. The technical capabilities are available and practical to use. There is more recognition of the value of the maintenance vs. the reactive approach to well care. However, extending these principles to domestic water wells remains an area needing improvement nationally and worldwide.

For Further Reading

Smith, S.A., and A.E. Comeskey. 2009. *Sustainable Wells: Maintenance, Problem Prevention, and Rehabilitation.* Taylor & Francis CRC Press, Boca Raton, Florida.

Technical articles on well maintenance and associated topics at *www.groundwaterscience.com*.

Groundwater: State of the Science and Practice

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