A Review of

Aquifer Storage Recovery Techniques

Prepared and submitted to the
Wisconsin Department of Natural Resources
by the Aquifer Storage Recovery
Technical Advisory Group
March 22, 2002

Executive Summary

Aquifer storage and recovery (ASR) involves injection of surface water or groundwater into an aquifer for storage and later extraction from the same well. This water management technique allows for the optimization of water treatment facilities and enables water utilities to provide for the periodic peaks in water demand without having to oversize other storage and distribution facilities. Annual operating costs for an ASR system are often somewhat higher than the cost of operating a conventional water treatment and distribution system. However, when comparing capital cost per unit of new capacity, development of an ASR system is often less costly than the development of other water supply and treatment alternatives.

Most municipalities in Wisconsin use local sources of water to meet their needs. When the quantity of local sources is unable to keep pace with demand, the municipality may be forced to look elsewhere. Several municipalities have built pipelines to Lake Michigan to supplement or replace local groundwater sources. As these municipalities anticipate the need for expanded treatment or storage capacity to meet peak demands for this water, they are considering ASR as an alternative. One ASR well has been developed and is undergoing pilot testing cycles in the City of Oak Creek. A second ASR well is currently under development in the City of Green Bay.

Full scale operation of these systems, and development of additional systems in the future in Wisconsin, will require 1) demonstrating that existing State and Federal regulatory requirements are satisfied, 2) waivers of these requirements on a case-by-case basis, or 3) statutory changes to the requirements for a specific class of application such as ASR. This report is intended to provide technical guidance to the Wisconsin Department of Natural Resources as it considers options for evaluating proposed ASR systems and for monitoring of such systems once they are operational.

Existing applicable regulations include the Federal Underground Injection Control Program Requirements, which prohibit injection of water that does not meet primary drinking water standards, the Federal Safe Drinking Water Act (SDWA), which requires that water recovered from an ASR well must meet primary drinking water standards before being put into the water distribution system, and the Wisconsin Groundwater Law, which prohibits practices that will lead to exceedances of groundwater quality standards at specified “points of compliance”. Although most maximum contaminant levels specified in the groundwater quality standards are identical to those of the SDWA, there are notable inconsistencies for certain chemicals that form as by-products of disinfection processes designed to remove microbial pathogens from drinking water.

Water that could be stored underground using an ASR system includes surface water and groundwater from another aquifer. Water intended for storage could be untreated, treated by filtration only, or treated by filtration and disinfection prior to injection. Treatment by filtration and disinfection prior to storage offers considerable advantages to utilities in terms of minimizing peak demands on treatment plants and allowing for development of ASR with relatively minor modifications to existing wells and the distribution system. This option would allow a utility to remove and/or inactivate pathogens prior to injection of the water into the aquifer. However, such a system would include the injection of disinfection by-products into the aquifer.

Injection and withdrawal of groundwater through ASR wells can change local and regional flow patterns and water levels in the aquifer used for storage. These changes can lead to increases or decreases in the amount of groundwater discharging to streams, lakes or wetlands. In some cases, there are changes in hydraulic parameters, which may affect well yield. Changes in flow patterns may also induce changes in water quality. Changes in groundwater quality induced by flow field changes as a result of ASR operations are subject to regulation under the Wisconsin Groundwater Law. Potential changes in aquifer dynamics need to be evaluated during preliminary design of ASR systems in order to estimate ASR recovery efficiency and to anticipate water quality or hydraulic impacts on other users of the aquifer. Geophysical logging, hydraulic testing, and numerical modeling are tools that can be used in this evaluation.

Subsurface storage of water that has a different chemical composition from that of native groundwater can induce geochemical changes to the aquifer solids as well as to the water itself. While changes in the overall water chemistry and mineralogy of an aquifer are generally small, subtle changes can have a disproportionately large effect on the trace elements. In particular, oxygen present in injected water or introduced to the aquifer as a result of changes in water levels and flow patterns may stimulate “redox” reactions that affect the solubility of metals and other sensitive elements. Some mobilized elements, such as arsenic, have the potential to create significant health risks. Others such as manganese, which are not considered significant public health concerns at regulated levels, may still reduce the suitability of the water for domestic use. Arsenic mobilization is of particular concern in Wisconsin because high arsenic concentrations already affect a number of wells in the Fox River Valley, where a lowering of water levels has exposed sulfide minerals to oxygen.

A full understanding of the potential for adverse chemical reactions to take place during ASR operations requires adequate geochemical modeling in conjunction with careful monitoring during pilot testing of ASR systems. If contaminants are produced or released at high concentrations by geochemical reactions, it is possible that reactions within small intervals could lead to exceedances of SDWA or groundwater law standards in the bulk water recovered from an ASR well. Even if pilot studies do not indicate that water recovered from the ASR well will violate water quality standards, utilities may still need to design monitoring systems to identify the presence of problem intervals in which SDWA or groundwater law standards might be exceeded in other wells as a result of ASR operations. Such monitoring systems could consist of multiple monitoring wells open only to relatively short intervals or single wells equipped with temporary packers or permanent multi-level sampling devices.

In addition to trace elements that might be mobilized by geochemical reactions, a variety of potential contaminants could be introduced to the aquifer by surface water, including microbial pathogens. Proper filtration and disinfection will remove or inactivate the majority of pathogens present in the source water. The DNR should consider requiring surface water utilities to meet all microbial filtration and disinfection standards set by the SDWA before the injection of surface water into the aquifer.

Herbicides and polychlorinated biphenyls (PCBs), inorganic chemicals like nitrite and nitrate, and metals like chromium and cadmium are also possible contaminants of concern. Although some regulated chemicals may be detected in some of Wisconsin's surface waters, none have shown up in Wisconsin's municipal surface water supplies at concentrations higher than standards set by the SDWA. Therefore, it is unlikely that such contaminants will be injected into an aquifer at concentrations that exceed health-based standards. In any case, these can be adequately controlled for public health protection if the injected water meets SDWA standards.

A number of chemicals that occur commonly as by-products of disinfection have been demonstrated to cause cancer and other adverse effects in laboratory animals at high doses. The processes responsible for generating disinfection by-products are designed to reduce the microbial load in a water system and prevent the transmission of waterborne diseases. Therefore, water utilities are faced with the competing goals of minimizing the risk of disease from microbial pathogens and keeping levels of potentially harmful disinfection by-products low. All of Wisconsin's municipal surface water utilities are well below the health-based SDWA standards for disinfection by-products. However, some of these substances can exceed Wisconsin's groundwater standards, leading to the possibility that injected water could violate the groundwater enforcement standards at the property boundary or some other specified point of compliance. The Wisconsin Legislature may want to consider a waiver of the enforcement standards for disinfection by-products in the case of ASR, based on consideration of relative risks of introduction of microbial pathogens to the aquifer and those associated with the disinfection by-products.

This report is respectfully submitted to the Wisconsin Department of Natural Resources.

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PART I
INTRODUCTION AND OVERVIEW

What is aquifer storage and recovery?

According to the most general definition, aquifer storage and recovery (ASR) involves "the storage of water in a suitable aquifer through a well during times when water is available, and recovery of the water from the same well during times when it is needed" (Pyne, 1995). As discussed in more detail in a later section of this report, this water management technique allows for the optimization of water treatment facilities. The technique also enhances water storage and distribution system capacity in a way that enables a water utility to provide for the periodic peaks in water demand that are often experienced during the operation of a water supply system without having to oversize other components of the water system.

The water stored in the aquifer during ASR operations may come from a variety of sources and may be subjected to varying degrees of treatment prior to storage and after recovery. For reasons discussed in a later section of this report, ASR applications for water utilities typically involve the injection of treated drinking water underground through a dual-purpose well (see Figure 1). The water is temporarily stored in the vicinity of the injection well in a suitable aquifer. As it is needed, the stored water is retrieved through the same well and recovered into the water distribution system with little additional treatment to comply with the water quality standards established for public drinking water.

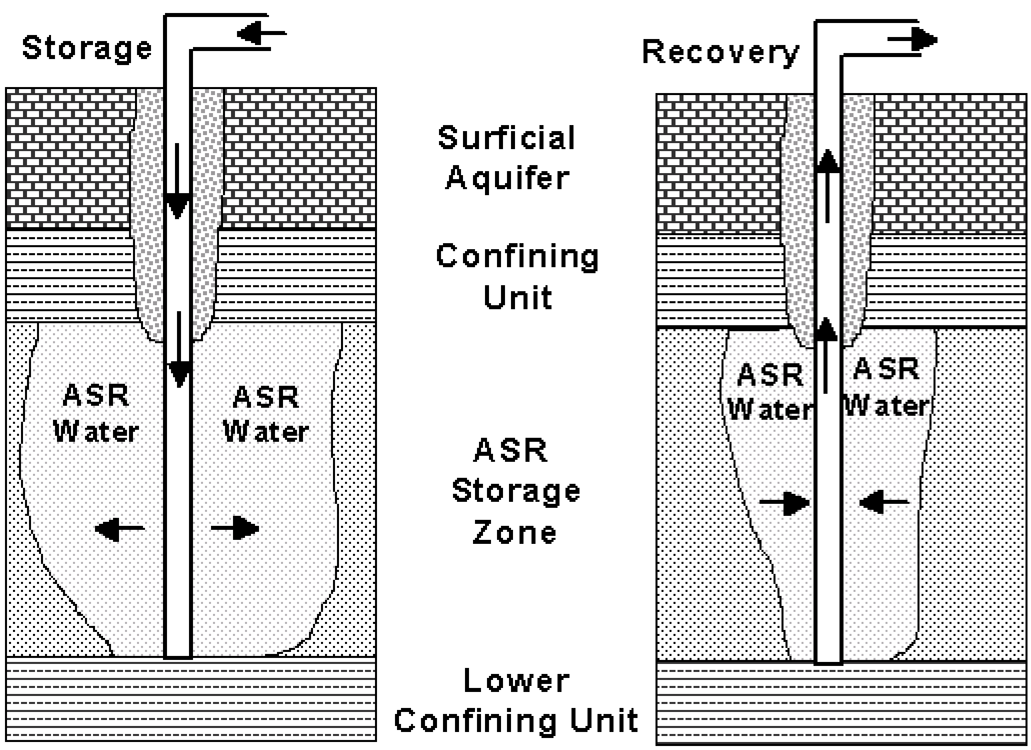

Figure 1. Schematic diagram of storage and recovery phases of ASR operation, with water stored in a confined aquifer. (After National Research Council, 2001a.)

The first ASR system to be used for seasonal storage and recovery of treated drinking water was developed in Wildwood, New Jersey in 1968. Since then, 20 additional ASR projects have become operational within the United States and another 40 projects are in various stages of development.

The first ASR well to be developed in the State of Wisconsin was completed in 1998 and is located in the City of Oak Creek. The well was approved by the Department of Natural Resources as part of a research study that was collaboratively funded by the Oak Creek Water and Sewer Utility and the American Water Works Association Research Foundation. A second ASR well is currently under development in the City of Green Bay.

ASR Applications Considered for this Report

ASR has been proposed or used for a variety of purposes. Many of these are described in Pyne et al. (1996) and they include

- seasonal storage of treated drinking water,
- storage of emergency supplies,
- use as an alternative to surface reservoirs for long term storage to meet demands during dry years,
- recharge with limited recovery to partially reverse the effects of long term water level declines due to overpumping,
- storage of reclaimed waste water that is intended for irrigation, and
- improvement of water quality through geochemical and microbiological reactions that are expected to occur in the subsurface.

Currently, the largest proposed ASR system in the U.S. is a regional network of approximately 200 wells near Lake Okeechobee in Florida. This regional system would be used to store excess surface water that would otherwise discharge to the ocean during times of peak flow. The stored water would be recovered during dry periods to maintain water flow through the Everglades.

Although a variety of applications could conceivably be proposed in Wisconsin, the primary application of ASR in this state is expected to be management of peak demands on water treatment plants. Hence, this report focuses primarily on issues associated with injection of treated drinking water for recovery into a water distribution system.

Issues related to who has the legal right to recover water that has been stored in an ASR system have been raised in other states. Water rights issues were not examined by the technical advisory group that prepared this report.

Considerations Motivating Utilities to Consider ASR as a Management Option

Water Availability and Cost

In the most general sense, water utilities must select water supplies that are capable of meeting

- water quality standards set by the federal and state government, and
- water quantity demands under peak conditions.

Available water supplies are often categorized into surface water sources and groundwater sources. For Wisconsin, the distributions of municipal utilities and population served among these two source types are shown in Table 1. Approximately 53% of Wisconsinites who are served by municipal water supplies receive water from a groundwater source.

Table 1. Wisconsin's municipal water supplies and the population they serve (data from WI DNR Drinking Water Data Base http://www.dnr.state.wi.us/org/water/dwg/DWS.htm)

Source	Number of Utilities [1,2]	Population Served
Lake Michigan	36	1,458,000
Lake Winnebago	5	177,000
Lake Superior	2	39,000
50% L. Winnebago + 50% Groundwater	2	10,000
Rainbow Lake	1	1,000
Surface Water	44	1,675,000
Groundwater	570	1,908,000
Surface and Groundwater	2	10,000

[1] The number of municipal water utilities is shown. There are another 11,159 non-municipal water suppliers serving a total population of 229,000 with groundwater. Individual homeowner wells are not included.

[2] The number of utilities includes those utilities that purchase their water from other utilities.

When selecting a water supply, utilities generally consider those sources having the lowest total cost. Examples of items that affect the total cost are proximity and quality of the source. The cost of systems needed to transport water increases as the distance from source to point of use increases. The cost of treatment systems increases as the quality of the water source decreases.

Most municipalities in Wisconsin use local sources of water to meet their needs. When the quantity of local sources is unable to keep pace with demand, the municipality may be forced to look elsewhere. If forced to look elsewhere, the municipality needs to negotiate agreements that will allow the municipality to collect either groundwater or surface water in some other jurisdiction and to build a pipeline across one or more jurisdictions. This may involve expensive political and legal battles over water rights and other issues (the recent experience with Perrier serves as an illustration of this). The cost of such legal battles is considered a part of the total cost of the water supply and may actually be a significant factor in which source ultimately gets selected. Green Bay went through this process in 1957, having built a pipeline from the city to

Lake Michigan after local groundwater supplies became insufficient. Oak Creek also turned to Lake Michigan in 1976 for the same reasons but, due to its location on the Lake Michigan shore, did not need to negotiate such agreements. The lack of sufficient groundwater supplies is now forcing suburban Green Bay communities to consider alternative water supplies.

Annual operating costs for an ASR system range from $6,000 to $40,000 per million gallons per day of recovery capacity (Pyne, 1995). This is often somewhat higher than the cost of operating a conventional water treatment and distribution system. However, when comparing capital cost per unit of new capacity, development of an ASR system is often less costly than the development of other water supply and treatment alternatives. Various estimates of capital cost savings range from 50 to 90% of the cost of conventional alternatives (Pyne, 1995).

Water Quality Issues

As noted above, utilities are required to comply with water quality regulations established by the federal and state governments. The DNR is responsible for enforcing these standards in Wisconsin. Utilities using surface water supplies are typically concerned with meeting standards for turbidity and for protozoan pathogens like *Cryptosporidium*. Lake Michigan and Lake Superior utilities have no difficulty meeting disinfection by-product standards while those using Lake Winnebago need specialized processes to meet these standards. Groundwater systems in Wisconsin have localized concerns with arsenic and radionuclides like radon and radium. These utilities also have widespread aesthetic concerns associated with hardness and localized aesthetic concerns with color, iron, and manganese. Forthcoming regulations may force groundwater utilities to implement treatment practices for viral pathogens. Some water supplies degrade in quality over time, as is currently happening at groundwater systems in both Brown (arsenic and gross alpha) and Waukesha (gross alpha and salinity) Counties. Under these circumstances, a utility will need to consider the implementation of new treatment practices and/or the development of a new water source.

Water Quantity Issues

Water utilities typically operate and maintain a source of supply, treatment facilities, and a distribution system. The distribution system consists of items like pipes, pumps, and storage tanks (e.g. water towers) that must be designed to handle the expected peak hour flow in the design year. The source and treatment facilities must normally be designed to meet the expected peak day flow in the design year. Typically, the peak day demand occurs sometime during the summer and the lowest demand occurs sometime during the winter. Therefore, treatment facilities are only used at or near their capacity during the summer of the design year. A significant amount of their capacity is not used during the winter, even in the design year.

If ASR is implemented and sufficient storage is available in the aquifer, a treatment facility can be used up to its capacity in the winter months with treated water being stored in the aquifer. During summer months, the utility can deliver more than its treatment capacity by recovering the water stored in the aquifer. Therefore, when ASR is used with sufficient storage, a utility can theoretically design a source and treatment system that meets average day demand in the design year rather than peak day demand. In practice, these facilities would likely be designed to meet a demand somewhere between the average day and peak day demands.

To give some perspective to these different demand numbers, the City of Oak Creek can be used as an example. Oak Creek's treatment plant is rated by the DNR as having the ability to produce 20 million gallons per day (mgd). In the year 2001, Oak Creek's average water production was 6.9 mgd and the highest production day was July 9, when 14.9 million gallons of water were produced. The ratio of peak day to average day was 2.14, which is higher than the range of 1.32 to 2.00 observed in Oak Creek from 1980 through 1999 (Miller, 2001). Therefore, the treatment plant had an unused capacity of 5.1 mgd on July 9 and an average unused capacity of 13.1 mgd.

Peak day demands at Oak Creek are estimated to exceed existing treatment capacity in 2010, implying that a new treatment plant expansion must be built in 2010 if ASR is not implemented (Miller, 2001). The average unused capacity in 2010 was estimated by Miller (2001) to be about 10 mgd, or 50% of the total capacity. Assuming the trends in average day demand can be extrapolated beyond those reported by Miller (2001), Oak Creek's average day demand would exceed treatment capacity in about 2070. Therefore, if these assumptions are correct and sufficient storage were available in an ASR system, Oak Creek could wait up to 60 additional years before needing a new treatment plant expansion.

The implementation of conservation programs can also delay the construction of new treatment facilities by lowering peak day demands. The period of the delay depends on how much conservation is currently practiced relative to the maximum amount of conservation that can be achieved. It is unknown whether such an analysis has been done for Oak Creek or Green Bay.

Regulatory Considerations Affecting ASR

Use of ASR systems in Wisconsin requires

- demonstrating that existing State and Federal regulatory requirements are satisfied,
- waivers of these requirements on a case-by-case basis, or
- statutory changes to the requirements for a specific class of application such as ASR.

A brief review of the relevant requirements and their relationship to ASR operations is provided below.

Underground Injection Control Program Requirements

In order to protect drinking water supplies from contamination, Part C of the Safe Drinking Water Act of 1974 requires that all underground injection practices be regulated. Provisions in the Code of Federal Regulations (40 CFR 144.12) prohibit the construction, operation, maintenance, conversion, plugging, or abandonment of any injection activity that allows the movement of fluid containing any contaminant into underground sources of drinking water, if the presence of the contaminant may cause a violation of any primary drinking water regulation under 40 CFR part 142 or may otherwise adversely affect the health of persons. In other words, preventing changes in the chemical and/or microbial composition of the groundwater is not the goal of the federal Underground Injection Control program. Rather, the goal is to prevent chemical and/or microbial changes that adversely affect the public health of future users of the groundwater.

ASR wells are considered to be a type of Class V injection practice. Any fluid that is injected underground via an ASR well must therefore comply with the federal primary drinking water standards. The United States Environmental Protection Agency (EPA) has established the point of injection as the place for determining compliance with the primary drinking water standards.

State and Federal Performance Standards for Water Recovery

In addition to meeting certain federal drinking water quality standards during the injection phase of ASR operations, any water that is recovered from an ASR well must also comply with federal drinking water quality standards before it can be legally reintroduced into any public water system. Currently, the water distributed within a municipal water system must comply with over 80 established contaminant standards.

The EPA establishes health-based and enforceable maximum contaminant levels (MCLs) for numerous chemicals and microbial pathogens under the authority of the Safe Drinking Water Act (SDWA). In general, MCLs established by the SDWA carry risks of adverse health outcomes in a range from 1 in 1,000,000 to 1 in 10,000. All state governments are required to adopt these SDWA standards at levels that are at least as stringent as those specified by the EPA. Wisconsin's version of these standards appears in Chapter NR 809 of the Wisconsin Administrative Code.

Most chemical and microbial contaminants are regulated at the entry point to the distribution system, suggesting that the water recovered from an ASR well must meet these standards prior to entering the distribution system. If the SDWA standards are not met by the recovered water, the ASR operation must be halted or the recovered water must be subjected to additional treatment before it enters the distribution system.

Other chemical and microbial contaminants are regulated at points in the distribution system because of their potential to increase in concentration at locations in the distribution system. Such contaminants include microorganisms like coliform bacteria and chemicals like disinfection by-products and corrosion by-products. Water recovered from the ASR well would need to be of sufficient quality to allow a utility to meet these standards.

Wisconsin Groundwater Law

In 1984, the Wisconsin Legislature enacted Chapter 160, Stats., which is more commonly referred to as the Wisconsin Groundwater Law. The intent of this legislation was to minimize the concentration of polluting substances in groundwater through the use of numerical water quality standards that are to apply to every state groundwater regulatory program. The standards developed under this authority are to be considered the criteria for the protection of public health and welfare. Once a standard is established for a contaminant, each state groundwater regulatory agency is required to review its administrative rules and, as necessary, revise those regulations to ensure compliance with the enforcement standards and preventive action limits established in Chapter NR 140, Wis. Admin. Code.

In many cases the groundwater enforcement standards are equal to the drinking water MCLs. However, there are a number of differences between the enforcement standards listed in NR 140 and the maximum contaminant levels listed in NR 809. Table 2 lists NR 140 standards that are more stringent than NR 809 standards while Table 3 shows NR 140 standards for which no NR 809 standard exists. This results in the possibility that treated drinking water intended for injection in an ASR well could meet all primary drinking water standards but would not comply with the groundwater quality standards.

Table 2
Substances for which the NR 140 standard is more stringent than the NR 809 standard

Substance	**NR 140 Enforcement Standard**	**NR 809 Maximum Contaminant Level**
Atrazine	3 µg/L *	3 µg/L
Bacteria, Total Coliforms	0	See NR 809.30 and NR 809.31
Bromodichloromethane	0.6 µg/L	80 µg/L for total trihalomethanes †
Bromoform	4.4 µg/L	80 µg/L for total trihalomethanes †
Chloroform	6 µg/L	80 µg/L for total trihalomethanes †
Dibromochloromethane	60 µg/L	80 µg/L for total trihalomethanes †
Polychlorinated biphenyls (PCBs)	0.03 µg/L	0.5 µg/L

* The NR 140 standard for atrazine applies to the sum of the concentrations of atrazine and the following microbial metabolites of atrazine: 2–chloro–4–amino–6–isopropylamino–s–triazine (formerly deethylatrazine), 2–chloro–4–amino–6–ethylamino–s–triazine (formerly deisopropylatrazine) and 2–chloro–4,6–diamino–s–triazine (formerly diaminoatrazine).

† The NR 809 standard for total trihalomethanes applies to the sum of the chloroform, bromodichloromethane, dibromochloromethane, and bromoform concentrations.

Table 3
Substances regulated by NR 140 but not by NR 809

Substance	NR 140 Enforcement Standard	Substance	NR 140 Enforcement Standard
Acetone	1000 µg/L	Ethylene glycol	7 mg/L
Aldicarb	10 µg/L	Fluoranthene	400 µg/L
Anthracene	3 mg/L	Fluorene	400 µg/L
Bentazon	300 µg/L	Fluorotrichloromethane	3.49 mg/L
Benzo(b)fluoranthene	0.2 µg/L	Formaldehyde	1000 µg/L
Boron	960 µg/L	N-Hexane	600 µg/L
Bromomethane	10 µg/L	Hydrogen sulfide	30 µg/L
Butylate	67 µg/L	Methanol	5 mg/L
Carbaryl	960 µg/L	Methyl ethyl ketone (MEK)	460 µg/L
Carbon disulfide	1000 µg/L	Methyl isobutyl ketone (MIBK)	500 µg/L
Chloramben	150 µg/L	Methyl tert-butyl ether (MTBE)	60 µg/L
Chloroethane	400 µg/L	Metolachlor	15 µg/L
Chloromethane	3 µg/L	Metribuzin	250 µg/L
Chrysene	0.2 µg/L	Naphthalene	40 µg/L
Cobalt	40 µg/L	N-Nitrosodiphenylamine	7 µg/L
Cyanazine	1 µg/L	Phenol	6 mg/L
Dacthal	4 mg/L	Prometon	90 µg/L
Dibutyl phthalate	100 µg/L	Pyrene	250 µg/L
Dicamba	300 µg/L	Pyridine	10 µg/L
1,3-Dichlorobenzene	1250 µg/L	Silver	50 µg/L
Dichlorodifluoromethane	1000 µg/L	1,1,1,2-Tetrachloroethane	70 µg/L
1,1-Dichloroethane	850 µg/L	1,1,2,2-Tetrachloroethane	0.2 µg/L
1,3-Dichloropropene (cis/trans)	0.2 µg/L	Tetrahydrofuran	50 µg/L
Dimethoate	2 µg/L	1,2,3-Trichloropropane	60 µg/L
2,4-Dinitrotoluene	0.05 µg/L	Trifluralin	7.5 µg/L
2,6-Dinitrotoluene	0.05 µg/L	Trimethylbenzenes *	480 µg/L
EPTC	250 µg/L	Vanadium	30 µg/L

* The trimethylbenzene standard is for the sum of 1,2,4-trimethylbenzene and 1,3,5-trimethylbenzene.

A notable example of the difference between NR 140 and NR 809 occurs in the case of the trihalomethanes, which consist of chloroform, bromodichloromethane, dibromochloromethane, and bromoform. The trihalomethanes are produced when chlorine is added as a disinfectant to kill microbial pathogens. Because they are produced during the disinfection process, the trihalomethanes are known as disinfection by-products. While the drinking water MCL for the combined concentrations of these compounds is 80 µg/L, a separate groundwater enforcement standard has been established for each compound individually. A typical water utility on Lake Michigan may produce chloroform concentrations of 10 to 25 µg/L and total trihalomethane concentrations of 15 to 40 µg/L. These concentrations are deemed safe by the NR 809 standard but unsafe by the NR 140 standard. Because NR 140 regulates some substances on a group basis (e.g. atrazine and PCBs), there is no clear reason why the trihalomethanes are regulated on an individual basis.

In addition to the above differences, it is also important to note that NR 809 standards will generally exceed NR 140 preventive action limits. In most cases, preventive action limits are set at values that are 10 to 20 percent of the NR 140 enforcement standards. According to Chapter 160, Statutes, "preventive action limits shall serve as a means to inform regulatory agencies of potential groundwater contamination problems, to establish the level of groundwater contamination at which regulatory agencies are required to commence efforts to control the contamination and to provide a basis for design and management practice criteria in administrative rules. A preventive action limit is not intended to be an absolute standard at which remedial action is always required."

Chapter 160, Stats., designates several specific points at which compliance with the groundwater quality standards must be attained. These points are

- any point of present groundwater use,
- any point beyond the property boundary of the premises where the facility, activity or practice is located or undertaken, or
- any point beyond the design management zone but within the property boundaries of the premises where the facility, activity or practice is located or undertaken.

No regulatory agency may promulgate an administrative rule that allows exceedance of a groundwater enforcement standard at a point of standards application.

Given the fact that municipal wells are often located on small plots of utility-owned land surrounded by property belonging to other public or private entities, it is likely that water injected through ASR wells will frequently migrate beyond utility property boundaries. If this injected water met drinking water standards for trihalomethanes but not the groundwater enforcement standards for individual disinfection by-products, migration of the injected water beyond the utility's property boundaries would violate the existing Wisconsin groundwater law.

PART II
PUBLIC HEALTH AND ENVIRONMENTAL CONSIDERATIONS

Source Waters to be Used in an ASR System

There are a number of water types that could be injected into an ASR well. These are summarized as follows:

- untreated surface water
- surface water treated with filtration but not with disinfection
- surface water treated with both filtration and disinfection
- untreated groundwater
- treated groundwater

Each of these possible scenarios has a unique set of advantages and disadvantages related to engineering.

Untreated Surface Water

Under this scenario, untreated surface water would be injected into the aquifer during periods of low demand. Stored water would be recovered during periods of high demand and processed in a treatment plant having both filtration and disinfection. With this option, the source water pumping station could conceivably be designed to meet average day demand rather than peak day demand. This represents a relatively small financial benefit but might be a significant advantage if the surface water supply were a stream having low flows in months of peak demand. Since no streams are used as Wisconsin municipal water supplies, this is currently irrelevant to the Wisconsin water industry. Although there are some savings in pump station costs, the treatment facility would need to be capable of meeting peak day demand. Because treatment plants are much more costly than the raw water pumping facility, there is little financial incentive to use this option. Due to the possibility of microbial contamination, this is also the least desirable option from a public health perspective.

In both Oak Creek and Green Bay, the ASR wells are connected to the distribution system. If this scenario were to be used at these locations, these two utilities would need to consider at least two options.

- Disconnect the existing well from the distribution system and install pipes to carry water from the source to the well and recovered water from the well back to the treatment plant.
- Install new wells at either near the source area or the treatment plant.

The costs of each option would typically be the controlling factor in deciding which to pursue. Because of these costs, it is plausible that the utilities might abandon the concept of using ASR if forced to use this scenario.

Surface Water Treated with Filtration but not with Disinfection

Under this scenario, untreated surface water would be processed through a filtration plant without disinfection and the filtered water would be injected into the aquifer during periods of low demand. This option would allow a utility to inject water having about 99% to 99.9999% fewer particles and pathogens than the untreated water. Such a system would also avoid the injection of disinfection by-products into the aquifer. Stored water would be recovered during periods of high demand and processed in a treatment plant having disinfection.

With this option, the source water pumping station and the filtration plant could conceivably be designed to meet average day demand rather than peak day demand. This represents a potentially significant cost savings although some significant engineering issues would need to be ironed out. Most treatment systems include disinfection either prior to or through the filtration process. For example, Green Bay employs ozonation prior to the filtration process in their existing treatment plant. To employ this scenario, Green Bay would need to make substantial hydraulic changes in the treatment system so that injected water does not receive ozonation while water sent to the distribution system receives ozonation.

As noted with the previous option, the ASR wells in Green Bay and Oak Creek are connected to the distribution system. If this scenario were to be used at these locations, these two utilities would need to consider at least two options.

- Disconnect the existing well or wells from the distribution system and install pipes to carry water from the filtration plant to the well and recovered water from the well back to the treatment plant for disinfection. It is plausible that disinfection could be performed at the well site if chlorine is used and sufficient contact time is provided. This would be unlikely in many cases due to the need for contact time.
- Install new wells at the treatment plant.

Again, the costs of each option would typically be the controlling factor in deciding which to pursue.

Surface Water Treated with both Filtration and Disinfection

This is the option that has been proposed for both Oak Creek and Green Bay. Under this scenario, untreated surface water would be filtered and disinfected at a treatment plant and the treated water would be injected into the aquifer during periods of low demand. In this case, treated water could be allowed to enter the distribution system prior to injection into the aquifer. This option would allow a utility to achieve 99.9% to 99.99999% removal and/or inactivation of pathogens prior to injection of the water into the aquifer. However, such a system would include the injection of disinfection by-products into the aquifer. Stored water would be recovered during periods of high demand and sent directly into the distribution system after the addition of disinfectant for residual maintenance.

With this option, all source water and treatment plant components could conceivably be designed to meet average day demand rather than peak day demand. Furthermore, if existing wells are hooked up to the distribution system, they can be used with relatively minor modifications in the ASR program. This option likely represents the most significant cost savings for surface water utilities from an engineering perspective.

Untreated Groundwater

From an engineering perspective, the issues associated with the injection of untreated groundwater are similar to those for untreated surface water, although the contaminants of concern are generally different (see later discussion of surface water contaminants versus groundwater contaminants). Under this scenario, untreated water would be pumped from one aquifer and then injected into a second aquifer. Water recovered from this second aquifer may require treatment before being sent to the distribution system. As with the surface water, the design of treatment facilities would need to be based on peak day demand. The cost implications of this depend on the level of treatment needed or desired.

Treated Groundwater

This option would involve the treatment of water from one aquifer followed by injection of the treated water into a second aquifer. Injection could occur at a well hooked up to the distribution system and recovered water could be placed directly into the distribution system. Savings could be gained on treatment facilities due to the ability to design for average day conditions. Again, the amount of savings would depend on the level of treatment.

ASR-Induced Changes in Aquifer Dynamics

General

Aquifer systems generally respond to development, either by withdrawal or injection of water, with changes in groundwater levels and discharge to surface-water bodies. In some cases, there are changes in hydraulic parameters, which may affect well yield. Changes in flow patterns may also induce changes in water quality. There are no state laws or regulations designed to prevent water quality degradation due to groundwater withdrawals. However, changes in groundwater quality induced by flow field changes as a result of ASR operations could be subject to regulation under the Wisconsin Groundwater Law.

Drawdown, Drawup and Well Interference. The water level in a well declines as water is pumped from the well. As pumping continues the water level (hydraulic head) in the well falls below that of the surrounding aquifer, which causes water to move from the aquifer to the well. This eventually causes a decline or drawdown in hydraulic head (groundwater level) in the aquifer, which will continue until the rate of flow into the well equals the amount of water supplied from the larger hydrologic system. Water flows to the well from the aquifer in all directions. The hydraulic gradient (the change in water level over a specified distance in a given direction) is steeper close to the well than farther away from the well, forming a cone of depression. When water is injected into a well as in the case of ASR, the opposite occurs; the water level in the well increases above the water level in the aquifer. Water flows from the well to the aquifer, an increase in aquifer hydraulic head (drawup) occurs and a cone of impression is formed.

When pumping or injection wells are located too close together they can interfere with each other, that is, pumping from one well can cause drawdown (or drawup in the case of injection) in another nearby well. The effects of these interferences can be added together (or superimposed) such that total drawdown or drawup is the sum of the drawdown or drawup caused by individual pumping or injection wells.

*Capture***.** Under pre-development conditions groundwater entering or recharging an aquifer system is more-or-less in equilibrium with groundwater discharge from the system. In Wisconsin, groundwater generally discharges to streams, lakes and wetlands. When groundwater withdrawal takes place, hydraulic gradients due to pumping wells change the groundwater-flow regime so that groundwater that would have discharged to surface-water bodies is captured by the pumping wells. Therefore, excessive pumping can reduce streamflows and drain wetlands.

Hydraulic parameters. Most municipal wells located in the Lower Fox River Valley and southeastern Wisconsin tap the sandstone aquifer. The sandstone aquifer in these areas is confined and includes many formations that have varying hydraulic properties. Some formations such as sandstone have a relatively high hydraulic conductivity (ability to transport water) while other formations such as dolomite or siltstone have a relatively low hydraulic conductivity. In places, dolomites may be fractured. Weathering may enlarge these fractures, giving rise to conduits for rapid groundwater flow. The amount of water that can be withdrawn from or injected into a well is dependent on the hydraulic conductivity of the formations tapped by the well and the saturated thickness of the formations. Transmissivity is the product of hydraulic conductivity and saturated thickness, and it, along with the storage properties of the aquifer, determines the aquifer productivity and possible well yield. Storage properties are largely dependent on the porosity of the formations comprising the aquifer.

In eastern Wisconsin the Maquoketa-Sinnipee confining unit confines the sandstone aquifer. If groundwater levels as measured in wells tapping the sandstone aquifer decline below the base of the Maquoketa-Sinnipee confining unit, the aquifer will have reduced water-bearing capacity because transmissivity will be reduced.

Case Histories Illustrating Potential Effects on Aquifer Dynamics

Two ASR pilot studies have been initiated in Wisconsin: the Green Bay pilot (Lower Fox River Valley) and the Oak Creek pilot (southeastern Wisconsin). In order to understand the possible effect of ASR on the groundwater system better, case studies that describe the hydrogeology in the vicinity of the ASR pilots are presented.

Lower Fox River Valley. The Lower Fox River Valley includes two pumping centers: the Green Bay metropolitan area and the Fox Cities area near the north shore of Lake Winnebago. Many studies have helped to define the groundwater resources of the Lower Fox River Valley and document the status of the groundwater system (Walker et al., 1998; Conlon, 1997; Batten and Bradbury, 1996; Consoer, Townsend & Associates, Inc., 1992; Emmons, 1987; Feinstein and

Anderson, 1987; Krohelski, 1986; Knowles, 1964; Knowles et al., 1964; LeRoux, 1957; Olcott, 1966; Drescher, 1953).
Most high-capacity wells in the Lower Fox River Valley obtain water from the sandstone aquifer. The sandstone aquifer comprises sedimentary rock beneath the Sinnipee Group (predominantly dolomite). Above the sandstone aquifer the Maquoketa Shale and Sinnipee Group form a confining unit. This confining unit is overlain by unlithified deposits and Silurian Dolomite, which form an upper aquifer. The Precambrian crystalline rock is the base of the active groundwater-flow system, because it is essentially impermeable.

In the upper aquifer, precipitation recharges groundwater in topographically high areas and movement is toward discharge areas such as streams and lakes in topographically low areas. Recharge to the sandstone aquifer occurs mainly to the west of the Lower Fox River Valley, where the Maquoketa-Sinnipee confining unit is absent and the sandstone aquifer is in good hydrologic connection to the upper aquifer. Groundwater movement in the sandstone aquifer prior to development was generally west to east. Since development, the direction of groundwater movement has been towards the Lower Fox River Valley near the Central Brown County and the Fox Cities pumping centers.

Pumping from closely spaced wells has resulted in large cones of depression in the vicinity of the pumping centers. These cones of depression have merged so that pumping in one area affects the other area thus making declining groundwater levels a regional problem. As early as 1953, researchers acknowledged that well interference was a problem in the Green Bay area, causing unneeded declines in water levels (Drescher, 1953). Since 1957, the City of Green Bay has used a pipeline for most of their water supply. The construction of the pipeline was prompted by excessive drawdown of the sandstone aquifer near Green Bay. During the last several years, a proposal to build an additional pipeline to Lake Michigan has been discussed by Central Brown County Water Authority, which includes representatives from communities in the vicinity of the Green Bay metropolitan area but does not include the city of Green Bay. Similar discussions have taken place in the Fox Cities; for example, the officers and directors of the Fox Cities Chamber of Commerce and Industry held a meeting in June 1994.

The Central Brown County Water Authority and East Central Wisconsin Regional Planning Commission provided estimates of municipal groundwater use for the year 2030. In the Central Brown County area, a 240 percent increase for the period 1990 to 2030 is projected, from 7.3 to 24.7 mgd. In the Fox Cities Heart-of-the-Valley communities (Combined Locks, Darboy, Kaukauna, Kimberly and Little Chute), a 41 percent increase for the period 1990 to 2030 is projected, from 3.9 to 5.5 mgd. For the Fox Cities western towns (Appleton, Greenville, Neenah, and Menasha), a 110 percent increase for the period 1990 to 2030 is projected, from 1.7 to 3.6 mgd.

Water-level measurements indicate there has been a steady decline in groundwater levels in the vicinity of these two pumping centers for at least the last 50 years. Figure 2 shows water levels measured in observation wells just to the north (BN-76) of the Green Bay metropolitan cone of depression and just to the south of the cone (BN-154). The rate of water-level decline in these wells is about 5 feet per year. Figure 2 also shows water levels from wells CA-6 and OU-326 which are located just to the north of the Fox Cities cone of depression and indicate a rate of water-level decline of about 2 feet per year. Even though the rate of decline is higher in the Green Bay metropolitan area than in the Fox Cities area, the saturated thickness of the sandstone aquifer and, therefore, its ability to produce water, is predicted to diminish in the vicinity of both pumping centers.

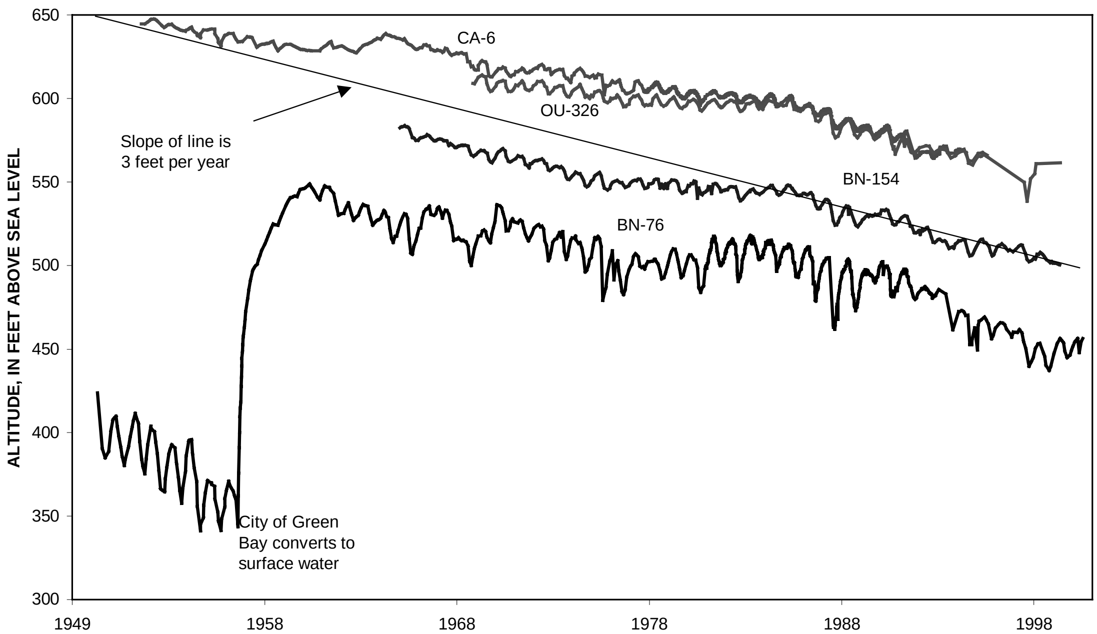

Figure 2. Sandstone aquifer groundwater levels measured in observation wells located in the Lower Fox River Valley

A simulation from a previously developed groundwater flow model (Conlon, 1997), using water-use projections for the year 2030, indicates water levels near the center of the cone of depression in the Green Bay metropolitan area will decline to almost the Precambrian bedrock surface. There will not be enough available drawdown to pump several wells located near the center of the cone. In the Fox Cities area, water levels in the year 2030 are predicted to drop about 50 feet below the top of the sandstone aquifer leaving only 500 feet of available drawdown. These declines will result in increased pumpage costs and a reduction in the amount of water that can be pumped from the sandstone aquifer, and may have unintended adverse effects on the water quality of the water pumped.

Southeastern Wisconsin. Southeastern Wisconsin (Washington, Ozaukee, Waukesha, Milwaukee, Racine, Kenosha and Walworth counties) is one of the most rapidly growing regions of the State. The economic growth and urban development, which in 1996 contained about 37 percent of the resident population of the State, have been due, in part, to the abundant water supply available for domestic and industrial uses. Lake Michigan is the source for about 70 percent of the water supply, mainly for the lakeshore counties of Ozaukee, Milwaukee, Racine and Kenosha, which are principally in the Great Lakes Basin. The counties of Washington, Waukesha, and Walworth (Mississippi River Basin) rely on groundwater for the vast majority of their needs because there are strict legal limits on the transfer of water from the Great Lakes basin to the Mississippi River Basin.

Similar to the Lower Fox River Valley, early researchers generalized the deep groundwater system in Southeastern Wisconsin into a “sandstone” aquifer composed of sedimentary rock confined by the Maquoketa Shale. Later researchers split the sandstone aquifer into several aquifers and confining units. Although several groundwater flow models have been developed, no attempt has been made to simulate the interaction of the shallow aquifers, consisting of dolomite and unlithified materials, with the deeper groundwater systems. Literature reviews of previous studies of the groundwater resources of Southeastern Wisconsin and Northeastern Illinois are available in Young et al. (1988) and Young (1992).

Assuming an increase in groundwater usage proportional to increases in population levels, groundwater usage could be expected to increase from about 90 mgd in 1995 to about 140 mgd by the year 2020. Water-level declines over the last 50 years range from 6 to 10 feet per year in the deep aquifer (Figure 3). Cones of depression centered on Waukesha and Chicago have intersected so that pumping in one area can affect water levels in the other area.

There is mounting evidence that pumping at high flow rates from wells tapping the sandstone aquifer in this area have caused adverse changes to groundwater quality. Analyses of groundwater from the three highest producing Waukesha Water Utility wells (Wells 5, 9 and 10) have indicated that total dissolved solids (TDS) concentration has been increasing since the mid-1980s (written communication, John Jansen, Aquifer Science and Technology, 2001). Based on a geophysical survey it appears likely that high TDS water occurs at depth in the sandstone aquifer and is being induced to flow to high capacity wells as increased groundwater development takes place (Jansen and Taylor, 2001). Gross alpha concentration is increasing in several southeastern Wisconsin groundwater supplies and is also thought to be related to increased groundwater development (Tim Grundl, written communication, University of Wisconsin-Milwaukee, Department of Geosciences, 2001).

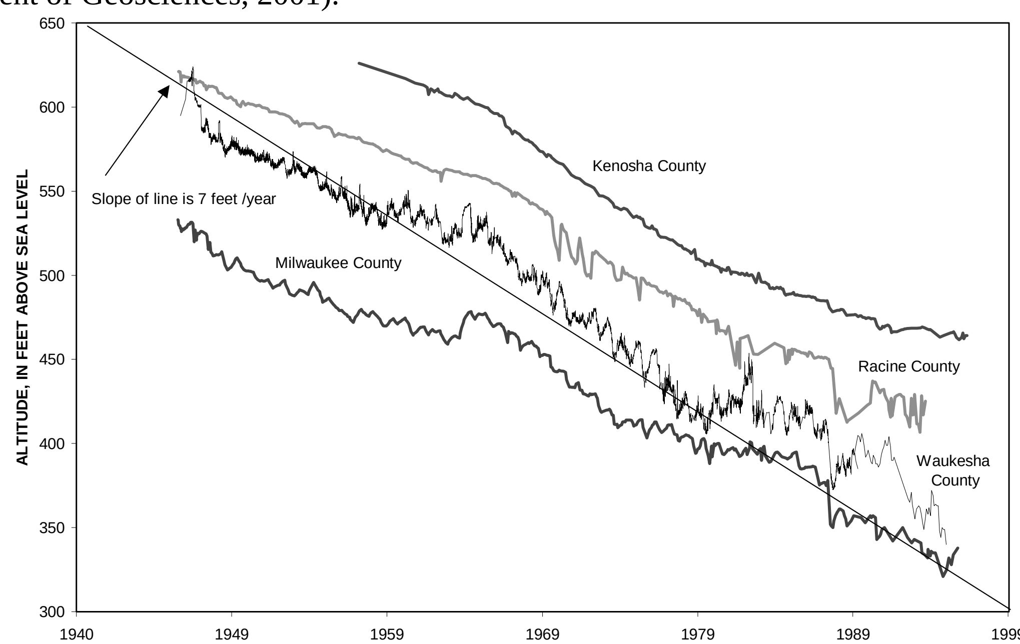

Figure 3. Sandstone aquifer groundwater levels measured in observation wells located in southeastern Wisconsin

ASR-Induced Changes in Aquifer Geochemistry

General

The process of ASR is, by definition, the injection of water from a different source that is likely to be of a different composition from the native groundwater. This process will inevitably lead to changes in aquifer geochemistry. Geochemical changes will occur to the aquifer solids as well as to the water itself. Changes in the overall (major-ion) aqueous chemistry and mineralogy of an aquifer are small, but subtle changes can have a disproportionately large effect on the trace element content of the groundwater. It is the trace element content that is of most importance for the maintenance of safe drinking water. Because ASR systems currently proposed for Wisconsin involve storage of surface water in the deep sandstone aquifer, this discussion focuses on changes in aquifer geochemistry that may take place in that type of ASR system.

The major-ion chemistry of surface waters in Wisconsin is typically dominated by calcium and bicarbonate. The sandstone aquifers in Wisconsin that are proposed for use in ASR contain water that is similar to surface water in the dominant major ions. These sandstone waters are typically dominated by calcium, magnesium and bicarbonate, although sulfate or chloride can be dominant in some areas of the state, notably Outagamie and Calumet counties. Geochemically, the most important differences between surface and groundwater are the higher dissolved oxygen and dissolved organic carbon content of surface water. Also important is the fact that the microbial consortium in surface waters is potentially different from the consortium native to groundwater systems. In general, surface waters are less concentrated than groundwater.

Trace elements. From a health and safety standpoint, changes in the major-ion chemistry of groundwater resulting from surface water injection are not of concern. However induced changes in trace element content caused by major ion differences can be problematic. It must be recognized that the chemistry of groundwater is dominated by reactions with the aquifer solids. This is particularly true for trace elements that reside almost entirely in or on the solid phases. When assessing potential effects caused by ASR systems, changes in the nature of aquifer solids as well as the changes in water composition must be considered. This is especially true when assessing the long-term health of the aquifer after ASR operations have been discontinued.

A change in the redox state of the system is the primary driving force in the potential mobilization of trace elements by ASR operations. Oxidative changes are caused by the addition of oxygenated surface water. Oxidative reactions can be controlled either abiotically or biotically. Reductive reactions are driven by the addition of organic carbon and are biotically driven. The interplay between oxidative/reductive changes induced by ASR and sorption/desorption or dissolution/precipitation reactions with aquifer solids (notably iron oxides and sulfide minerals) can be complex. A full understanding of this interplay requires adequate geochemical modeling in conjunction with careful monitoring.

Oxidative mobilization of trace elements commonly arises from the oxidation of iron sulfide minerals contained within the aquifer. A wide variety of iron sulfide minerals exists, all of which are subject to oxidative reactions. Iron sulfide minerals all contain significant amounts of trace metals and metalloids (eg. arsenic and selenium). The most common of these sulfide minerals is pyrite. Oxidation of pyrite releases sulfide and other metalloids as mobile oxyanions (eg. sulfate, arsenate and selenate). Iron and other metals are released as cations. In an oxidized system, the released iron precipitates as any of a suite of ferric hydroxide minerals, all of which are highly sorptive phases. During precipitation, ferric hydroxides can also accept trace elements into their lattices and, as a result, trace elements that originally resided in sulfide minerals can be found sorbed or co-precipitated to ferric oxides. If sufficient manganese is present, manganese hydroxide minerals will also precipitate and serve as additional sites for sorption and co-precipitation.

Reductive mobilization of trace elements arises from the reductive dissolution of iron and manganese hydroxides and the subsequent release of sorbed or co-precipitated trace constituents into the groundwater. This reductive mobilization can be coupled to oxidation of dissolved organic carbon introduced with the surface water. Pilot tests of the Oak Creek ASR system (Miller, 2001; CH2M Hill, 2001) have shown significant increases in dissolved manganese and iron in the observation well during the storage phase, which may be the result of reductive dissolution of iron and manganese hydroxides in the aquifer. Manganese and iron concentrations also showed a steady increase at the ASR well during the recovery phase of each pilot cycle. Manganese exceeded the drinking water standard at the end of recovery in cycles 1 through 4 but not in the most recent cycle, number 5. Increases in manganese concentrations resulting from introduction of organic carbon have been observed in other well fields in North America and Europe where pumping induces infiltration from nearby surface water bodies (Thomas et al., 1994)

Trace constituents that are sorbed to ferric oxides or other mineral surfaces are not bound permanently to the surface but can be released by dissolution of the surface, as mentioned above, by changes in the pH, or by increased competition from other dissolved constituents. The presence of carbonate minerals in the aquifer buffers the pH of deep sandstone water and pH levels typically remain between 6.5 and 8.5. It is in this pH range that the surface charge of ferric oxides changes from positive to negative. The sorption of oxyanions is sharply reduced once the surface becomes negative. This effect, along with competition with the hydroxyl ion (OH^-) causes a general decrease in anion sorption as conditions become more alkaline. The inverse is true for cation sorption (i.e. the extent of sorption decreases as conditions become more acidic), but this effect occurs at pH levels well below pH 6.5.

Sorption of a given ion will be affected by the presence of competing ions in solution. An increase in salinity will favor desorption of previously sorbed species. This is particularly true for species that display similar geochemical behavior. A well-known example of competitive sorption occurs between phosphate and arsenate.

Microbes. Unexpected biogeochemical processes can be initiated either by the addition of a new consortium of microbes or the stimulation of indigenous microbes caused by the addition of oxygen, carbon or other limiting nutrients. To avoid introducing a new microbial consortium it is important to inject disinfected water into the subsurface. Any stimulation of the indigenous population is detectable by proper monitoring of the water chemistry. Redox sensitive changes in water chemistry are useful for this purpose. Examples could include a rise in dissolved iron and manganese, a loss of dissolved organic carbon, a loss of sulfate, an increase in sulfide, an increase in alkalinity, or the presence of measurable hydrogen gas.

Changes After Cessation of ASR Operations. Long-term quality of the groundwater produced by the aquifer after ASR operations cease will be affected by the return of more reduced and concentrated groundwater. Reductive dissolution of iron and manganese oxides and increased competition for sorption sites are possible mechanisms to mobilize trace constituents. This is particularly true if previous ASR operations have caused the sequestration of trace elements on the surfaces of aquifer solids. If sufficient organic carbon is present, it is possible for redox conditions to become low enough that sulfate reduction is reached. Under these conditions, other oxyanions present (e.g. arsenate, selenate, and molybdate) will be reduced and sequestered, along with any trace metals present, in sulfide minerals or co-precipitated with pyrite.

Arsenic in the Fox Valley. An illustrative example of how specific geochemical reactions can cause deterioration in groundwater quality is provided by examining the causes behind the high arsenic concentrations observed in wells of the Fox River Valley.

High arsenic concentrations have been measured in many wells located in the Fox River Valley of Winnebago, Outagamie and Brown Counties. Concentrations as high as 12,000 μg/L are observed (Schreiber et al., 2000). The apparent source of arsenic to these wells is a naturally occurring layer of sulfide-rich cement. This "sulfide cement horizon" (SCH) occurs below the base of the Platteville Formation (Simo et al., 1996). The Platteville Formation is a group of dolostone and argillaceous dolostone units that occurs near the top of the sediments that comprise the deep sandstone aquifer. This SCH consists mainly of arsenic-bearing pyrite and marcasite crystals (pyrite and marcasite are both iron sulfide minerals). The source of the SCH is likely secondary hydrothermal brines that migrated out of the Michigan Basin through the more permeable St. Peter sandstone (Simo et al., 1996). The arsenic content is variable, but can reach 1% to 2% by weight (Simo et al., 1996).

Normal groundwaters contain only small amounts of dissolved oxygen, and under these conditions, pyrite and marcasite are very insoluble minerals. However pyrite and marcasite are subject to oxidative dissolution if oxygen levels rise. It has become clear that in the Fox River Valley, arsenic contained in the SCH is mobilized (as arsenate, AsO_4^{3-}) into the adjacent groundwater only in areas where the SCH intersects the regional water table or a local water surface in a well. When the pyrite contained in the SCH lies at a water surface, it is exposed to elevated levels of oxygen, begins to dissolve and release arsenic. The clear correlation between wells with high arsenic concentrations and proximity of the SCH to the water table is shown in Figure 4 (Dave Johnson, personal communication, 2002).

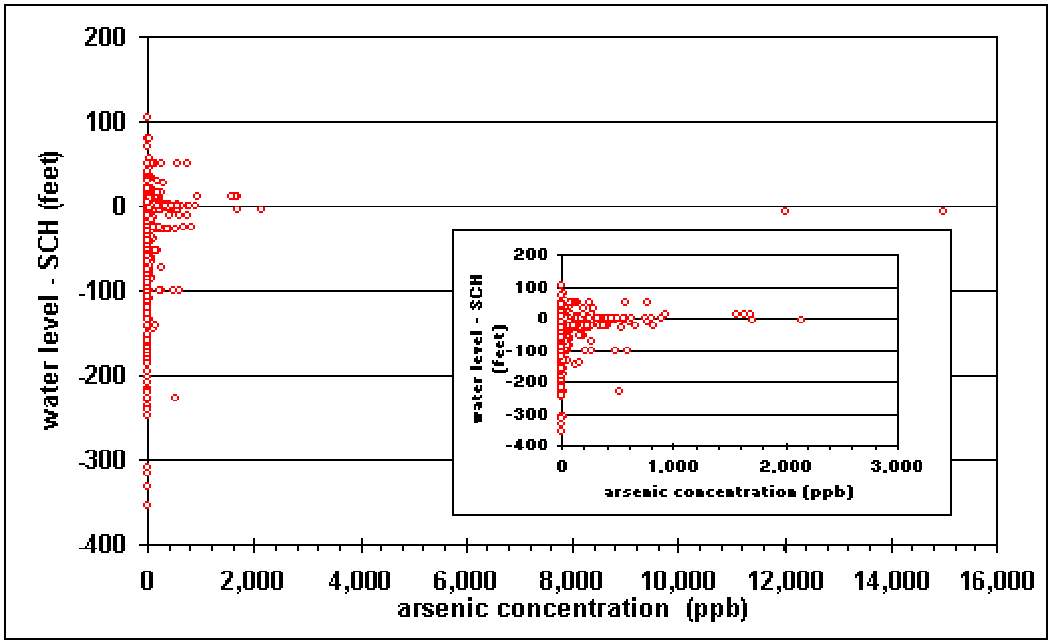

Figure 4: Arsenic concentrations versus proximity of water level and sulfide cement horizon (SCH). Proximity is depicted as the difference between water level elevation and SCH elevation, with negative values indicating that the water table is below the SCH. Plot is a compilation of data from Brown, Outagamie, Shawano and Winnebago counties. Inset shows that the trend continues at lower concentrations.

In areas where the water table is at or below the SCH, the well bore itself provides a direct conduit for oxygen to come in contact with arsenic-rich pyrite. (Schreiber et al., 2000) Another probable source of dissolved oxygen is from percolating recharge water. Recharge water in this area contains as much as 2.4 mg/L dissolved oxygen (Batten and Bradbury, 1996). To illustrate the fact that trace elements reside mostly on aquifer solids, not in the groundwater, mass balance calculations were made based on the regional water chemistry and the volume percent sulfides in the SCH (assumed to be 5%). These calculations indicate that oxidation of 1% of the sulfide minerals within a radial distance of 0.03 cm outside the borehole would be sufficient to provide the arsenic levels observed in standing water of contaminated wells. (Schreiber et al., 2000).

Although in this region arsenic is not an anthropogenic contaminant, but rather a naturally occurring element, the mobilization of arsenic is strongly influenced by human-induced changes to the aquifer. As seen in Figure 2, continued pumpage in the Fox River Valley is lowering water levels at a rate of approximately 3 feet per year. As water levels continue to decline, more of the SCH will inevitably become exposed leading to further arsenic mobilization. The potential for pyrite dissolution – and arsenic mobilization - also exists when fully oxygenated surface water (typical dissolved oxygen levels of 9 mg/L) is injected into the deep sandstone aquifer. It is for this reason that a clear understanding of the aquifer mineralogy, groundwater flow patterns and geochemistry must be obtained before ASR operations are started.

Potential Contaminants and Health Effects

General

From the public health perspective, there are a number of issues to be considered before the implementation of ASR at any site. First, there is a potential for contaminants that may be in the source of supply to be introduced into the aquifer during storage. This is a concern for both existing and future users of the aquifer. Second, contaminants that may be present in the groundwater could be distributed to utility customers during recovery of stored water. This would expose the utility customers to contaminants that they had not been previously exposed to. Finally, geochemical reactions caused by the introduction of non-native water into the aquifer may release contaminants into the aquifer.

These issues are complicated by the fact that the concentrations of some chemicals in the aquifer will decrease with the injection of non-native water while the concentrations of other chemicals will increase. For example, it is conceivable that the introduction of surface water to an aquifer may increase trihalomethane concentrations (due to continued reaction of chlorine with dissolved organics or aquifer organic carbon in the subsurface). At the same time, mixing of non-native water with native groundwater could decrease concentrations of naturally occurring groundwater contaminants such as radium. Under this situation, the DNR and the utility may need to conduct a risk assessment to determine whether the increased risk associated with the trihalomethanes is offset by the decreased risk associated with the radium.

When Surface Water is Introduced to an Aquifer

Several contaminants can conceivably be introduced to the aquifer by surface water. These include particulates like clay, silt, algae, and microbial pathogens. Dissolved substances like synthetic organic chemicals, inorganic chemicals, and metals are also possible contaminants of concern.

<u>Particles and Pathogens</u>. Untreated surface water may contain a range of pathogenic microbes including bacteria, viruses and parasites. These microbes may enter a water source from animal waste or sewage. Diseases commonly associated with microbial pathogens in drinking water include gastroenteritis, giardiasis, cryptosporidiosis, shigellosis and hepatitis. The most common bacterial contaminants found in drinking water are coliform bacteria. While not pathogenic, coliform bacteria provide an indication that a water system has been compromised and that pathogenic microbes may be present as well.

The contaminants likely to be of most significant public health concern for Wisconsin's utility customers are microbial pathogens like *Cryptosporidium*, which are present in surface waters like Lake Michigan on an occasional basis. Although some microbial pathogens will not survive several months of storage in the aquifer, it is well established that *Cryptosporidium* oocysts can survive for this period of time under such conditions (Fayer, 1997). Many particles will probably be concentrated in the aquifer near the injection site. However, a possible concern for other groundwater users in the vicinity of the ASR well might be the transport of microbial pathogens through the aquifer. Transport and fate will depend on the physical and chemical characteristics of the geologic formation, the physical and chemical characteristics of the pathogens, and the survival ability of the pathogens. However, there are insufficient studies available to determine what the extent of this problem might be in ASR systems. To help answer questions such as this, the DNR is encouraged to participate in the funding of research projects that monitor ASR systems over an operational time period of several years.

The probability for aquifer contamination by pathogens is greatly reduced if surface water is treated by both filtration and disinfection prior to storage in the aquifer. Proper filtration and disinfection will remove and/or inactivate 99.9% to 99.9999% of the pathogens present in the source water. Therefore, the DNR should consider requiring surface water utilities to meet all microbial filtration and disinfection standards set by the Safe Drinking Water Act (SDWA) before the injection of surface water into the aquifer.

As noted above, particles would likely be concentrated in the aquifer near the well, resulting in a slug of particulate material being pulled up from the well during the initial recovery phase (this has been observed in the Oak Creek pilot studies). In practice, this slug of particulate material can be sent to waste until some turbidity and/or particle count criteria are met. The DNR may wish to consider establishing such criteria for the recovery phase of all ASR operations.

Dissolved Chemicals. There are numerous chemical contaminants of health concern that are regulated under the SDWA. These include synthetic organic chemicals like herbicides and polychlorinated biphenyls (PCBs), inorganic chemicals like nitrite and nitrate, and metals like chromium and cadmium. Although some of these regulated chemicals may be detected in some of Wisconsin's surface waters, none have shown up in Wisconsin's municipal surface water supplies at concentrations higher than MCLs set by the SDWA. Therefore, it is unlikely that such contaminants will be injected into an aquifer at concentrations that exceed federal health-based standards.

Once injected into the ground, the fate and transport of these chemicals needs to be considered. Some chemicals are mobile and will migrate into the surrounding aquifer where they will be diluted by mixing with the native water, resulting in lower concentrations than in the injected water. Other chemicals will be microbiologically degraded in the aquifer, also resulting in lower concentrations than in the injected water. Although some locations in the vicinity of the injection well may see an increase in the concentration of such substances, the concentration will always be less than the health-based standards set by the SDWA. Therefore, chemicals meeting these characteristics can be adequately controlled for public health protection if the injected water meets SDWA standards.

Those chemicals with a strong affinity for the soil or rock in the aquifer may accumulate if they are not biodegraded or diluted quickly enough, possibly producing a higher concentration than in the injected water after some period of time. Such a scenario could be a concern for the utility using ASR and for other nearby users of the groundwater supply. So far, no research has been done to demonstrate whether or not this occurs in ASR systems, particularly at the low concentrations established by the SDWA. Again, the DNR is encouraged to help provide funds for answering important research questions like these that will be of benefit to the entire state.

Disinfection By-products. Disinfection by-products consist of organic and inorganic substances produced by the interaction of chemical disinfectants with naturally occurring substances in the water source. For example, trihalomethanes and haloacetic acids are formed by the interaction between natural organic matter and chlorine. Bromate is produced when ozone reacts with bromide and bromoform (a trihalomethane) is produced when ozone reacts with both bromide and natural organic matter. The processes responsible for generating disinfection by-products are designed to reduce the microbial load in a water system and prevent the transmission of waterborne diseases. Therefore, water utilities are faced with the competing goals of minimizing the risk of disease from microbial pathogens and keeping levels of potentially harmful disinfection by-products low.

A number of chemicals that occur commonly as by-products of disinfection have been demonstrated to cause cancer and other adverse effects in laboratory animals at high doses. Findings from epidemiology studies in populations exposed to disinfection by-products from treated surface water have provided some confirmatory evidence for excesses of some cancers (most notably bladder and colon cancer) and for several adverse pregnancy outcomes, including miscarriage, birth defects and low birth weight (Health Canada, 1999).

All of Wisconsin’s municipal surface water utilities are well below the health-based SDWA standards for disinfection by-products. However, some of these substances can exceed Wisconsin’s groundwater standards. As with the chemicals described in the previous section, the fate and transport of these by-products in the aquifer also needs to be considered. Studies at Oak Creek and elsewhere have generally shown that the trihalomethanes decrease in concentration during the recovery phase, and possibly during the later storage periods, of an ASR operation. While these decreases have been attributed to in-situ degradation of these compounds, the decreases may also be due to dilution by mixing with native groundwater. Studies outside of Wisconsin have shown similar decreases during recovery for the haloacetic acids. Whatever the mechanism responsible for decreases in concentrations at Oak Creek, total trihalomethane concentrations have been below health-based SDWA standards at all times. However, the concentrations of some individual compounds have been higher than Wisconsin’s groundwater standards for much of the storage period. To date, no studies have evaluated the fate and transport of bromate in ASR systems.

Substances Mobilized by Geochemical Interactions. Surface waters have many natural constituents that are not considered detrimental to public health. However, when introduced into an aquifer for the first time, some of these chemicals may initiate geochemical or microbiological reactions that release substances of public health concern. For example, dissolved oxygen will be introduced into the groundwater environment when surface water is injected. Addition of oxygen to a previously anaerobic groundwater system may also stimulate microbially mediated redox reactions, which can affect the speciation and solubility of metals and other redox sensitive elements. As discussed in the previous section, the fate and transport of released elements will depend on the chemical characteristics of the geologic formation as well as on the chemical characteristics of the water. Some mobilized elements, such as arsenic, have the potential to create significant health risks. Others such as manganese, that are not considered significant public health concerns at regulated levels, may still reduce the suitability of the water for domestic use. Health and other concerns associated with these two elements are discussed below.

Arsenic. Arsenic is an inorganic metallic element that occurs naturally in groundwater in Wisconsin, most notably in Brown, Outagamie and Winnebago Counties. The EPA has proposed a new MCL of 10 μg/L. The proposal to reduce the MCL from 50 μg/L to 10 μg/L was based on a growing body of research in recent decades showing increased occurrence of cancers of the skin, lung and bladder in human populations at levels below the existing MCL. The most exhaustive reviews of toxicological and epidemiological data for arsenic in drinking water have been conducted by panels assembled by the National Research Council (NRC).

In its 1999 report, a NRC panel found that data from populations exposed to arsenic at levels above 100 μg/L in Argentina, Chile and Taiwan were sufficient to establish definitively that exposure to arsenic in drinking water is associated with increased incidence of cancers of the skin, lung and bladder (NRC, 1999). Chronic non-cancer effects seen in human populations exposed to arsenic include hyperpigmentation and keratosis of the skin, peripheral vascular disease, hypertension, and onset of diabetes mellitus. Studies examining reproductive and developmental effects in humans have yielded mixed findings. Based on its risk model, the panel derived a point estimate for excess bladder cancer risk of 1 to 1.5 cases per 1000 at the level of 50 μg/L, and concluded that at the existing MCL, the risk of cancer was higher than EPA's target cancer risk range of 1 in 10,000 to 1 in 1,000,000 for MCLs.

At the request of EPA, the panel was reconvened to include in its analysis data that were not available prior to its 1999 report. The panel concluded that analysis of the newer data resulted in an enhanced confidence in the link between exposure to arsenic in drinking water and cancer, and that point estimates for excess lung and bladder cancer risk were in the range of 1 to 7 cases per 1000 at the proposed standard of 10 μg/L – a higher estimate than in the 1999 report (NRC, 2001b).

Manganese. Manganese is a metallic element that is commonly present at low levels in Wisconsin groundwater. When present in household water supplies, manganese can cause the staining of plumbing fixtures and laundry. Both Wisconsin and EPA have established secondary drinking water regulations for manganese at 50 μg/L based on public welfare concerns not related to health.

Manganese is an essential dietary element, necessary for bone formation and the metabolism of amino acids, lipids and carbohydrates. The NRC has established levels for adequate intake at 1.8 mg/day for females and 2.3 mg/day for males, and has established a tolerable upper intake level at 11 mg/day (NRC, 2001c). Conversely, exposure to manganese has also been linked to neurological impairment. EPA has established a reference dose of 0.14 mg/kg/day for manganese based on deficits in neurological function tests in human populations exposed to high levels of manganese (1.6 – 2.3 mg/L) in drinking water (Kondakis et al., 1990). A reference dose is an estimate of a daily exposure to the human population that is considered likely to be without an appreciable risk of deleterious effects during a lifetime. Furthermore, EPA included in its assessment a modifying factor of 3 to be applied to reduce the reference dose when used for drinking water because of concerns about the increased bioavailability of manganese in drinking water and the proximity of the level at which effects were observed in the Kondakis study and the level of essentiality.

When Groundwater is Introduced

Microbial pathogens are not likely to be a significant issue when water from one aquifer is stored in another. The greatest likelihood of a pathogen problem would occur if water from a shallow well were injected into another aquifer. Microbes may enter shallow aquifers from septic system effluent or from wells that are improperly constructed or sealed. The presence of dissolved chemicals having a detrimental effect on public health would depend on the location. In this state, naturally occurring arsenic and radionuclides are likely to be the most significant concerns. Some dissolved chemicals may also cause some aesthetic concerns (those that concern the taste, smell, or visual appearance of the water). These would include salinity, hardness, iron, manganese, and color. Disinfection by-products may be a concern primarily in groundwater systems with high amounts of organic color (such as Wausau) or with high salinities. As in the case of surface water introduction, geochemical reactions could be initiated if the blending of water from another aquifer causes significant changes in pH, ionic strength (e.g. salinity), or redox state.

PART III
ASR-RELATED RECOMMENDATIONS

Groundwater Modeling to Evaluate Potential Changes in Aquifer Dynamics

Numerical groundwater flow modeling can be used during the conceptual design phase of an ASR system to evaluate the potential effects on the groundwater flow field. Calibrated groundwater flow models can be used to analyze and simulate the effects of various design considerations such as site selection, well configurations, and pumping schedules on ASR system performance. For example, models can simulate the recovery that may occur due to reduction in pumping and injection. These changes may affect the pattern of groundwater flow and transport. Modeling can also be used in the early stages of ASR projects to guide the collection of field data and to test conceptual models of groundwater systems.

Groundwater flow models can be regional or site scale. Regional models can provide boundary conditions for site-scale models, which, in turn, can be used to evaluate pumping or injection strategies for individual wells or at a resolution not afforded by regional models. Site-scale models (telescopic mesh refinements) based on regional models ensure that the far-field sources and sinks of water are simulated correctly (Anderson and Woessner, 1992). In general, regional models are appropriate for simulation of regional stress (e.g. simulation of the effects of pumping from a pumping center) and site-scale models are appropriate for simulating local stress (e.g. simulation of the effects of pumping from an individual well).

A regional groundwater flow model is available for the area encompassing the Green Bay ASR pilot in the Lower Fox River Valley (Conlon, 1997) and a groundwater flow model will be available for the area encompassing the Oak Creek pilot in Southeast Wisconsin (U.S. Geological Survey, Wisconsin Geological and Natural History and Southeastern Wisconsin Regional Planning Commission, in progress). These groundwater flow models are three-dimensional finite-difference MODFLOW (McDonald and Harbaugh, 1988; Harbaugh and McDonald, 1996) models. The grid resolution of the Lower Fox River Valley model in the vicinity of the Green Bay ASR is 1320 feet on a side (40 acres) and the Southeast Wisconsin model grid is 2500 feet on a side (143 acres). The southeast Wisconsin regional flow model will be completed by Spring 2003.

As a demonstration of the usefulness of a regional model, ASR was simulated in the Green Bay area using the Lower Fox River Valley model.

Simulation of the Effects of ASR in the Green Bay Area

In the first simulation the Lower Fox River Valley model was run from pre-development to the year 2000. All of the high capacity wells shown on Figure 5 are pumping at reported 2000 rates. The pumping results in the formation of a large cone of depression in the Green Bay area. In the next simulation the municipal wells in the Green Bay metropolitan area (total of 41 wells) are used for ASR (Figures 6 and 7). All other municipal and industrial wells (total 83 wells) throughout the Lower Fox River Valley and the Fond du Lac area were pumped at 2000 rates.

This second simulation was started using the 2000 water levels from the first simulation as a starting point.

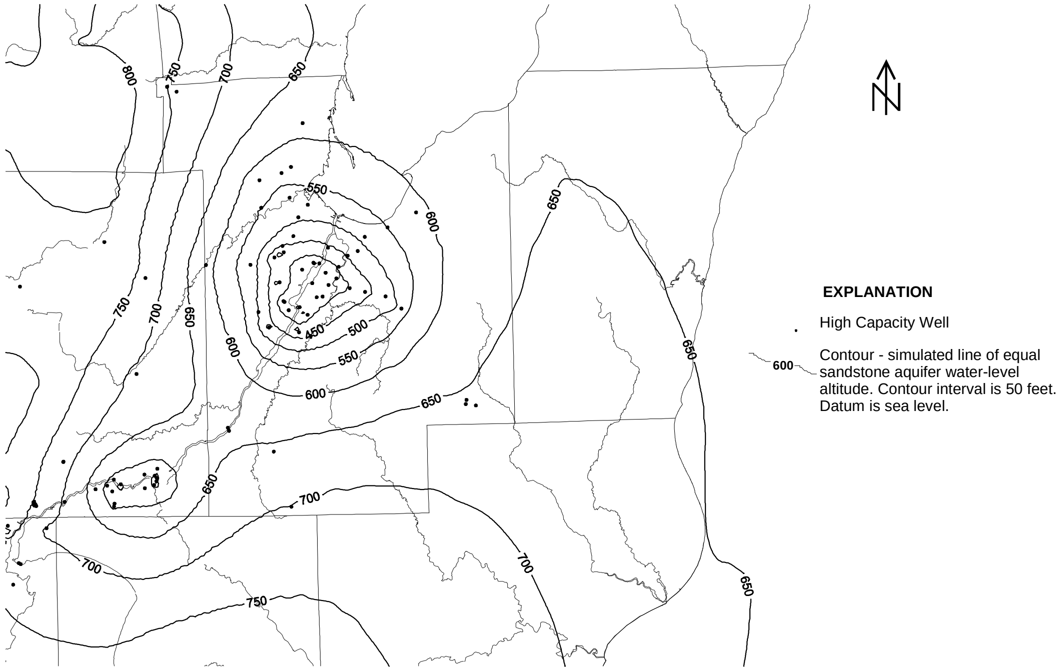

Figure 5. Simulated groundwater levels in the sandstone aquifer, 2000.

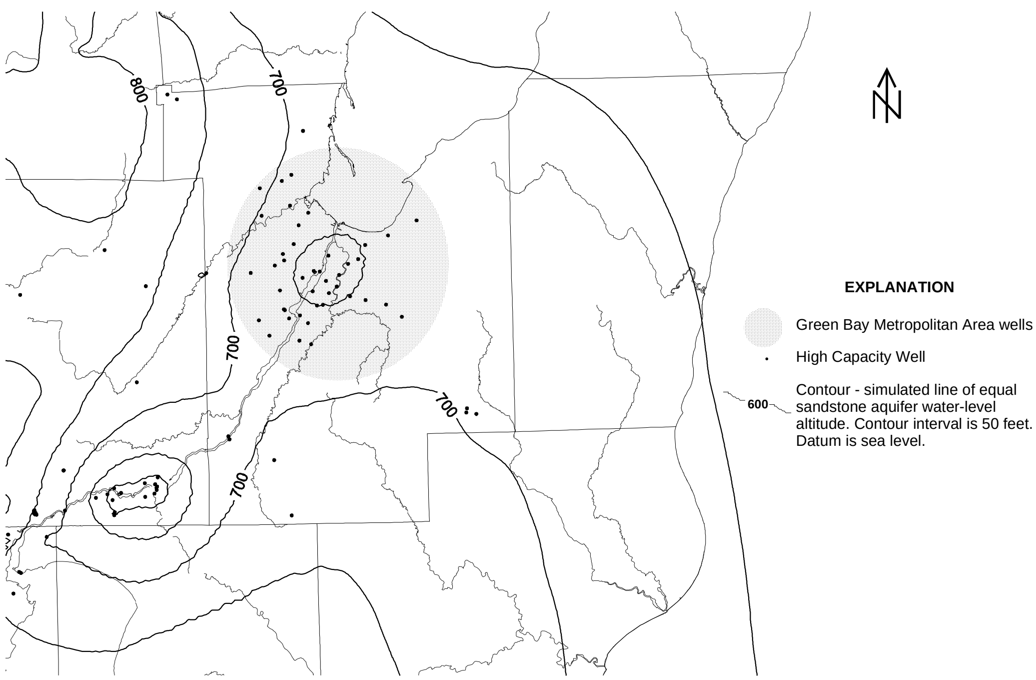

Figure 6. Simulated water levels in the sandstone aquifer, 2000, due to ASR. All of the municipal wells in Green Bay Metropolitan Area (total of 41 wells) were used for ASR.

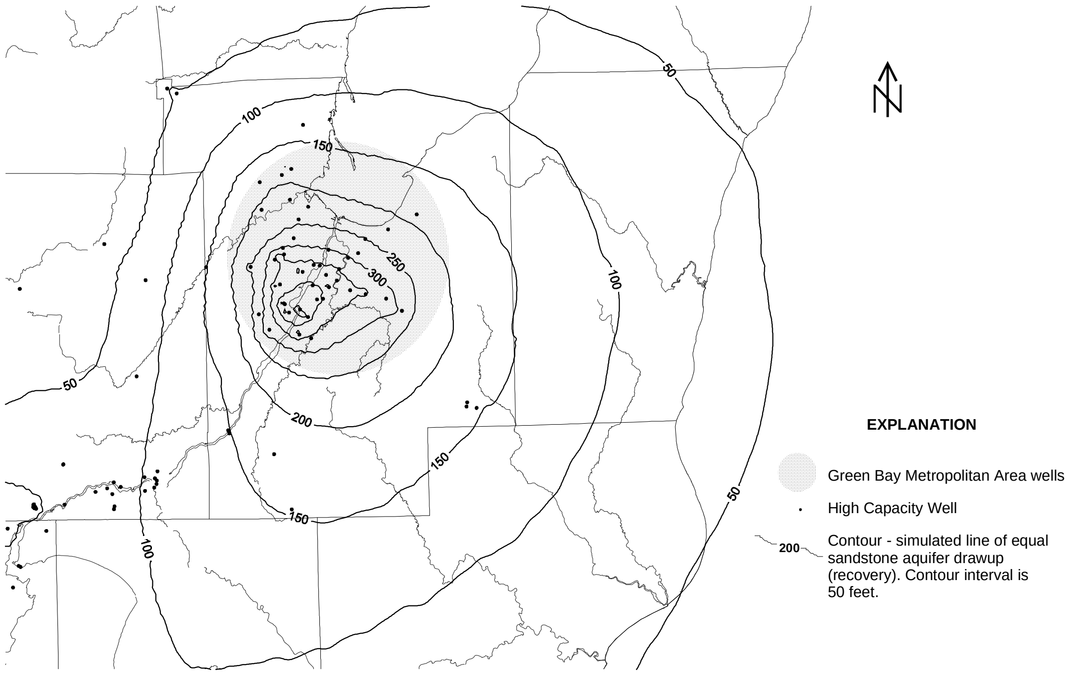

Figure 7. Simulated drawup (recovery) in the sandstone aquifer, 2000, due to ASR. All of the municipal wells in Green Bay Metropolitan Area (total of 41 wells) were used for ASR.

ASR was simulated for one year using four stress periods. Stress period 1 was 90 days and 50,000,000 gallons of water were injected into each of the 41 wells. Stress period 2 was 60 days and the 41 wells were not pumped or injected. Stress period 3 was 60 days and 50,000,000 gallons of water were pumped from each of the 41 wells. Stress period 4 was 150 days in which the 41 wells were not pumped or injected. As expected, simulation 2 results (Figure 7) indicate that the combined effect of the cessation of pumping the 41 Green Bay area wells and employing ASR causes a large drawup (recovery) in sandstone aquifer water levels. Water levels located near the center of the former cone of depression have recovered more than 400 feet. Transient forward particle tracking, in which groundwater flow paths are identified by following the movement of hypothetical particles in the simulated flow field, was conducted near a well located close to the center of the cone and used for ASR. The tracking was done by placing 10 particles of water at each of 3 depth locations in a circular pattern around the outer edge of the cell that contained the pumping well. The particles moved a maximum of 10 feet during injection (stress period 1) and then back 6 feet during pumping (stress period 3) for a net movement of 4 feet away from the well during the one-year of simulation. This indicates that all of the water injected was not recovered. A previous model run for a simulation time of 100 years without ASR indicated the total travel distance near a pumped well in the Green Bay area is about 2000 feet (Dave Saad, USGS, oral communication, 2001) which means water may move about 20 feet in one year without ASR.

A third simulation was performed using only the city of Green Bay wells for ASR, because it is unlikely that all of the Green Bay area municipal wells would be used for ASR (Figure 8).

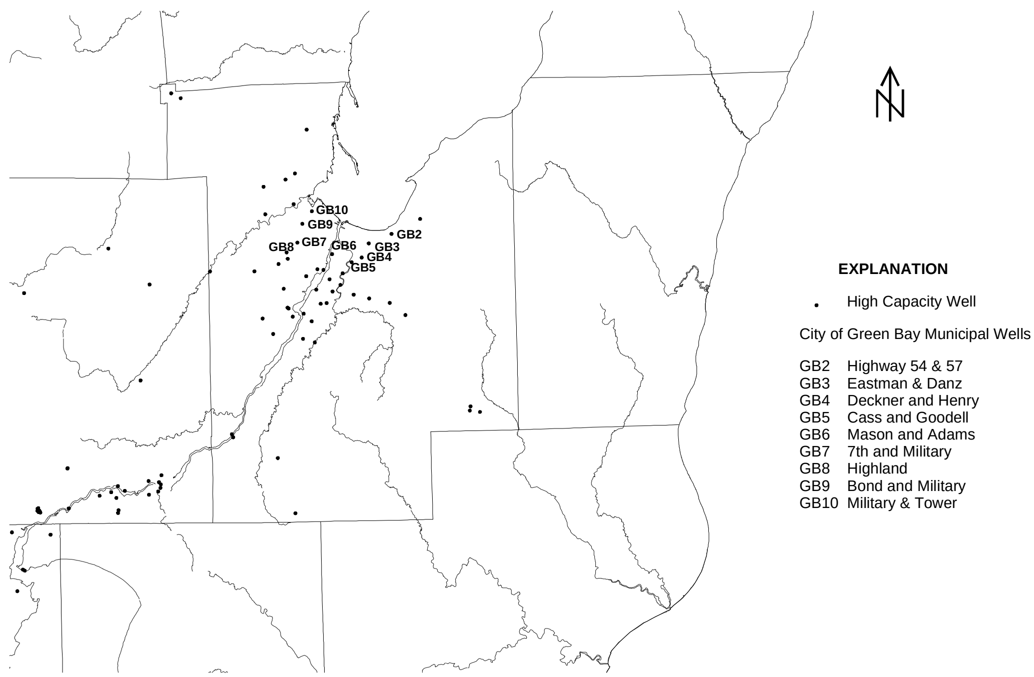

Figure 8. Locations of high-capacity wells and Green Bay municipal wells. Green Bay wells identified in the explanation were used to simulate ASR.

Water levels from the first simulation are used to start this simulation. The four stress periods covering one year described for the previous simulation were used again and, as before, all of the municipal and industrial wells, except the city of Green Bay wells, pump at year 2000 rates. Figure 9 is a graph that shows simulated daily water levels in the vicinity of Green Bay well 10 and Green Bay well 6 with and without ASR for a one-year period. For the first stress period when water is injected, and during the second stress period, when there is no injection, water levels are as much as 100 ft higher than they would be without ASR. During the third stress period when water is withdrawn the opposite occurs and water levels are as much as 100 ft lower than they would be without ASR. The water levels recover by the end of the fourth stress period almost to where they were at the beginning of the simulation.

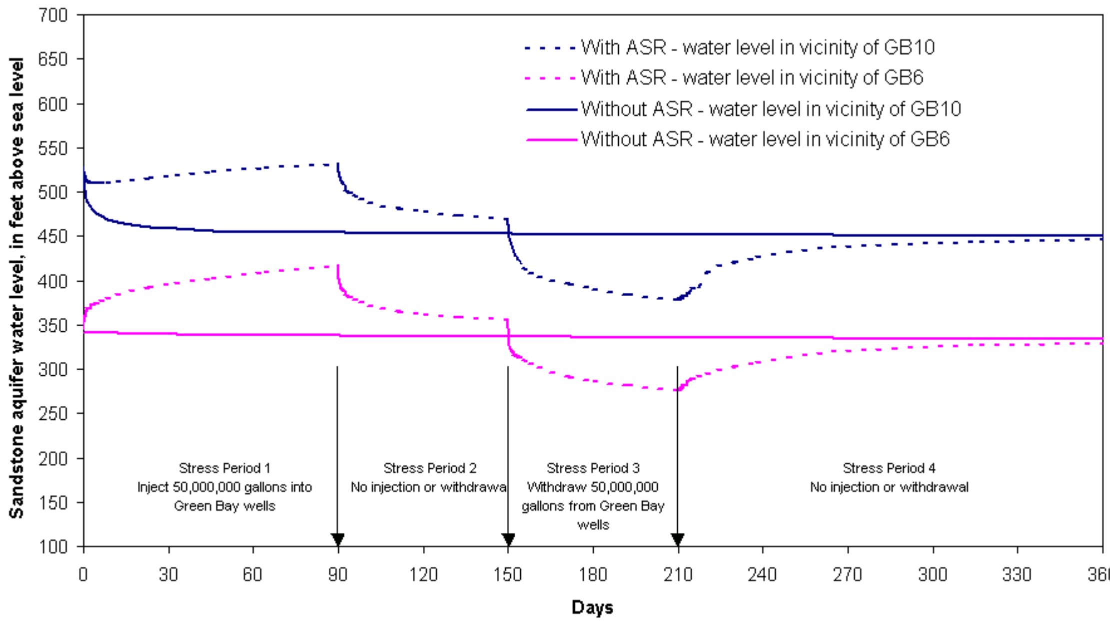

Figure 9. Simulated sandstone aquifer water levels at two locations with and without Green Bay municipal wells being used for ASR. Locations of Green Bay municipal wells (GB10 and GB6) are shown on Figure 8.

Need for Local Scale Models

It must be noted that the Lower Fox River Valley groundwater flow model, and regional models in general, are primarily useful to simulate general effects of ASR on aquifer systems because they are based on a preliminary conceptualization of a groundwater system. A conceptualization is an essential first step because it forms the framework for model development. The conceptualization reduces the real world into important component parts. This reduction is a necessary simplification of the natural system because inclusion of all of the complexities of the natural system into a computer model is not feasible. Steps in the development of the conceptual model include 1) definition of aquifers and confining units, 2) identification of sources and sinks of water and, 3) identification and delineation of hydrologic boundaries present in the area of interest. The intended use (regional versus site specific) of the model often dictates the degree of simplification that takes place during conceptualization of a groundwater system.

Regional models often have coarser grid spacing than site-scale models. Computer resources can limit model resolution; both cell size and number of aquifers simulated contribute to the amount of simulation time and computer memory required. The grid resolution of a regional model is generally not adequate to simulate the effect of a single small well on the groundwater system or a small area. Regional models often lump geologic formations into a limited number of aquifers with uniform hydraulic properties (both vertical and horizontal) applied over large (regional) areas. In addition, heterogeneities, fractures and preferential pathways for water are usually not simulated in regional models.

Determination of the effect of ASR on a groundwater system requires the application of a site-scale model in which all of the important hydrostratigraphic units and heterogeneities are simulated. The Lower Fox River Valley regional model is not suitable for this determination because the sandstone aquifer is treated as a single aquifer even though it is known that the Elk Mound Group has a higher hydraulic conductivity than the overlying St. Peter Sandstone. Water injected into high-capacity wells in Green Bay will probably flow at a faster rate in the Elk Mound Group than it would in the St. Peter Sandstone, which will not be discerned by the regional model. These differences are expected to affect calculated drawups and drawdowns, as well as the particle track distances and travel times and estimated amounts of injected water recaptured during the withdrawal phase of ASR.

Summary

The above analysis provided an illustration of how numerical groundwater flow modeling can be used during the conceptual design phase of an ASR system to evaluate the potential effects on the groundwater flow field. As noted earlier, such modeling can help the DNR and the utility make the following decisions:

- Where to locate an ASR well or set of ASR wells,
- How to select screened or open intervals for an ASR well or for each ASR well in a set of ASR wells,
- What injection and recovery schedules are needed for an ASR well or each ASR well in a set of ASR wells,
- Where to collect monitoring data for pilot tests and for long-term operation.

Construction of ASR Wells and Monitoring of Pilot Tests

Wells used for ASR may be newly constructed wells that are designed specifically for injection and recovery or they may consist of existing production wells that are retrofitted to allow ASR operations. Construction of a new well provides the best opportunity to select open intervals that may minimize adverse changes to aquifer dynamics and geochemistry while maximizing the efficient recovery of stored water. Retrofitting of existing wells may provide only limited flexibility to optimize efficiency and minimize adverse impacts, but this option generally offers significant advantages to a water utility in terms of construction costs. The ASR systems that are currently in development or proposed for Oak Creek and Green Bay make use of existing municipal wells.

Whether the proposed ASR well is an existing well or one that will be drilled and constructed specifically for ASR operations, an understanding of the hydraulic and geochemical properties of the units to which it will be open is essential to assessing potential hydrodynamic and geochemical impacts. Potential effects of hydraulic and geochemical heterogeneity are discussed below along with recommended methods that can be used to quantify these types of heterogeneity during design and pilot testing of ASR systems.

Hydraulic Heterogeneity

Although simple conceptual models of migration of injected water away from the ASR well often portray this as a relatively symmetric and uniform process, variations in hydraulic conductivity within the injection zone can result in significant variations in the extent of the "bubble" as a function of depth as illustrated schematically in Figure 10.

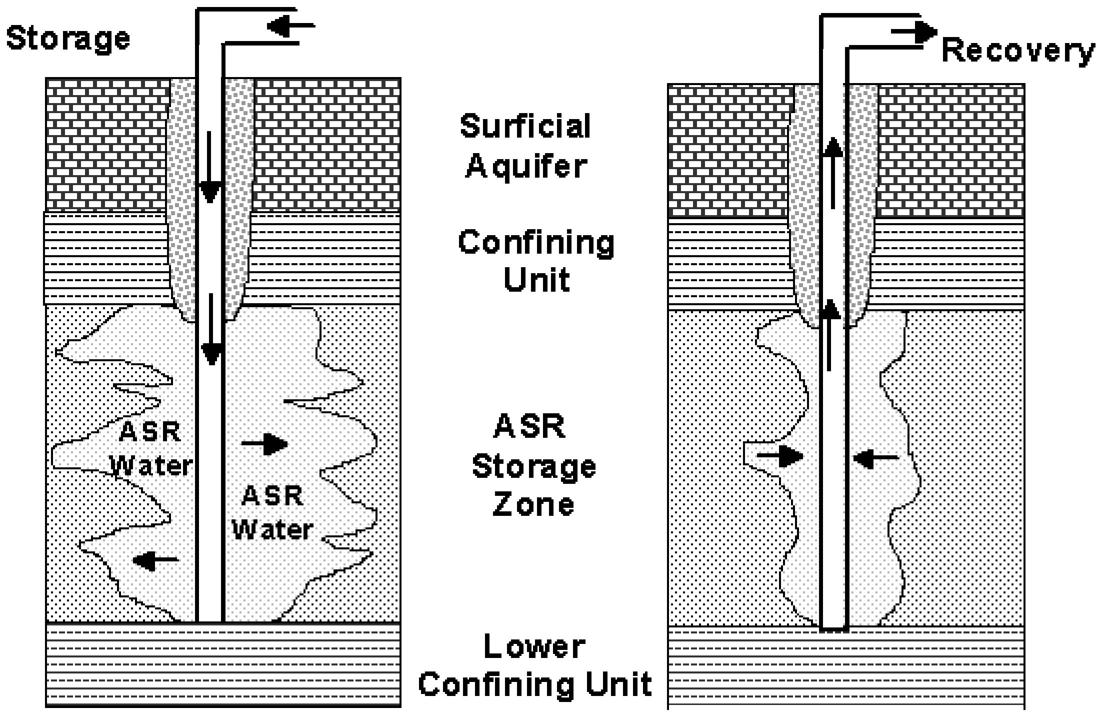

Figure 10. Schematic diagram of storage and recovery phases of ASR operation illustrating irregular bubble formation as a result of heterogeneity in the storage zone.

There have been few, if any, careful studies of existing ASR systems to assess the relationship between hydraulic heterogeneity in the injection interval and efficiency of recovery of the injected water. The following discussion is based on general hydrogeologic principles. Hydraulic variability can result not only in significant variations in bubble diameter during injection, but it can also affect the rates of continued migration of the bubble away from the ASR well under the ambient groundwater flow field during the storage phase. Zones of high hydraulic conductivity, which will preferentially accept the injected water, are also likely to be the zones in which flow rates are highest under ambient conditions. Although these high hydraulic conductivity zones would be expected to contribute preferentially to the well during the recovery phase of ASR operations, they may contain a greater proportion of native groundwater mixed with the injected water because of the greater migration during storage. Poor recovery efficiency has been observed in some ASR wells open to zones of very high hydraulic conductivity (Reese, 2000; Pyne, 1995).

Significant variations in hydraulic conductivity with depth occur in most of the sedimentary bedrock units in Wisconsin. Short-interval packer tests and down-hole flow meter tests by

Muldoon and co-workers (2001) in Door County wells open to the Silurian dolomite demonstrate a five order of magnitude range in hydraulic conductivity. That study identified high hydraulic conductivity preferential flow zones that can be correlated on the basis of geophysical logs and that extend over distances up to 25 miles. Stocks (1998) used a similar set of techniques to quantify hydraulic variations and identify laterally continuous flow zones in the Ordovician Sinnipee Group dolomites of eastern Wisconsin. Hydraulic conductivity variations over three orders of magnitude and the existence of laterally extensive, geophysically correlatable preferential flow zones have also recently been identified in some of the Cambrian sandstone units in Dane County (Swanson, 2001; Bahr and Parent, 2001).

Given the widespread potential for high hydraulic conductivity preferential flow zones in bedrock aquifers of the state, hydrostratigraphic characterization for ASR systems should include logging and other tests that can be used to identify such zones and to assess their potential effects on bubble migration and recovery efficiency. In the case of newly constructed ASR and monitoring wells, sampling of cores or cuttings combined with downhole geophysical logging can be used to identify lithologies and stratigraphic units. In areas where previous studies have related geophysical signatures to preferential flow paths, interpretation of the geophysical logs can aid in preliminary identification of zones for hydraulic testing. Borehole flowmeter studies under ambient and pumping conditions should be conducted to confirm the presence or absence of preferential flow zones. Flowmeters should be selected with consideration of expected flow rates and variations in flow rates. In some cases, it may be necessary to employ several different types of flow meters (e.g. spinner and heat pulse) in order to quantify contributions of both high and low hydraulic conductivity zones. Short-interval packer tests and pumping tests employing packed off intervals in the production and/or monitoring wells also can provide information on variations in hydraulic conductivity. Selection of appropriate zones for packer placement is an important factor in the usefulness of data from these tests for constraining the locations of preferential flow zones.

Once the hydraulic properties of the target injection zone and adjacent stratigraphic units have been characterized, ASR well design must consider a variety of trade-offs between long and short screens (or uncased open intervals). Compared to a well with a short open interval confined to a single hydrostratigraphic unit, one with a long open interval is likely to allow injection under lower pressures and with less perturbation of water levels in the vicinity of the well. However, a long open interval is also more likely to result in a highly irregular bubble. It may then be difficult to monitor bubble migration away from the ASR well during injection or to quantify the percentage of injected water that can ultimately be pumped back out of the aquifer during the recovery phase. A single monitoring well open over the same long interval as the injection well will yield water samples that reflect mixing of water from zones containing only native groundwater and those containing injected water. Tracking the migration of the most rapidly expanding portions of the bubble would require sampling from shorter packed-off intervals or a nested set of monitoring wells open to specific zones of expected preferential flow. Difficulty in monitoring will also affect the potential to anticipate adverse geochemical impacts (as discussed in the following section) and to demonstrate compliance with groundwater enforcement standards for contaminants such as disinfection by-products and microbial pathogens. Long or short open intervals that include high hydraulic conductivity preferential flow zones will not only preclude simple monitoring of bubble migration, but may also limit recovery efficiency.

The existing production wells in Oak Creek and Green Bay that have been proposed for conversion to ASR wells have open intervals that are hundreds of feet long. While it may be expensive to retrofit these to reduce the open interval or isolate the well from preferential flow zones that could limit recovery efficiency, it is still important to identify variations in hydraulic conductivity and preferential flow zones, particularly if these have implications for design of pilot test or operational monitoring.

Geochemical Heterogeneity

Variations in aquifer mineralogy and in groundwater chemistry are important controls on the potential geochemical changes that can be induced by injection of water with a different composition. As discussed in the section on ASR-induced changes in aquifer geochemistry, injection of water of a chemical composition that is different from the native groundwater has the potential to mobilize trace elements. Identifying aquifer zones that are potential sources or sinks for these elements is important to evaluating the potential for ASR operations to generate undesirable concentrations both in water recovered from the ASR well and in water that remains in the aquifer, possibly to migrate away from the immediate zone of ASR storage.

Trace elements of concern are unlikely to be evenly distributed throughout bedrock aquifers. As one example, the arsenic in bedrock aquifers of eastern Wisconsin is primarily associated with a sulfide bearing cement horizon located below the base of the Sinnippee Group (Schreiber et al., 2000). However, even within this horizon, arsenic concentrations of the sulfide minerals are quite variable. Geochemical analyses of core material collected as part of the Green Bay ASR project also revealed considerable variation in bulk arsenic concentrations with depth, with the highest arsenic concentrations found in core samples that contained no visible sulfide minerals (CH2M Hill, 2000). Water samples collected from a monitoring well during packer tests for the Green Bay ASR project contained arsenic concentrations of 45 μg/L when the sampled interval was open to the zone of highest arsenic detection in core samples (CH2M Hill, 2000). In contrast, water samples collected from intervals that were not open to this zone had a maximum arsenic concentration of only 6.3 μg/L. Significant variations as a function of sampling interval were also observed in concentrations of other dissolved species including manganese, sulfate, and radon. Historical records indicated average arsenic concentrations of 3 μg/L for nearby Municipal Well 10 (which is open over the entire producing interval), but a 24 hour constant rate flow test conducted as part of the ASR characterization yielded samples with concentrations of over 60 μg/L.

While a number of hypotheses can be proposed to explain the variations in concentrations of arsenic and other dissolved species as a function of depth and time, careful testing of these hypotheses is necessary to build confidence that the correct mechanisms are incorporated into any predictive models. Concentrations measured in samples from a monitoring well during cycle testing provide one means of hypothesis testing. However, interpretation of monitoring results may be limited if the concentrations are the result of mixing of waters from many intervals with distinct composition. Furthermore, any geochemical modeling of potential changes in arsenic mobility during ASR operations should be based on solute concentrations that represent the intervals of interest, rather than concentrations that are the result of mixing of water from different portions of the aquifer.

Design and construction of monitoring wells for evaluation of potential adverse geochemical changes must take into account variability in both aquifer geochemistry and hydraulic properties. Nested monitoring wells open only to relatively short intervals of interest identified by initial hydraulic and geochemical investigations provide one strategy for obtaining appropriate samples for monitoring and interpreting water chemistry changes during cycle testing. Alternatively, temporary packers or more permanent multi-level sampling devices can be used in a single well to allow sampling from isolated intervals. Results of geochemical analyses and modeling conducted as part of pilot tests may dictate modifications to the final construction and operation of the ASR well. For example, in cases of potential arsenic mobilization it may be desirable to seal off selected zones of the ASR well to limit adverse geochemical reactions.

Summary

Hydraulic and geochemical heterogeneity can lead to vertical variations in concentrations of chemical substances considered to be contaminants. Although the source water may have a uniform composition over the period of injection, once it enters the aquifer, geochemical reactions induced by mixing with native water or by contact with the aquifer solids can lead to vertical variations both away from and near the ASR well. In many cases contaminants originating in discrete intervals would be diluted with uncontaminated water produced from other intervals. But if contaminants are produced or released at high concentrations by geochemical reactions, it is possible that reactions within small intervals could lead to exceedances of SDWA or groundwater law standards in the bulk water recovered from an ASR well.

From the perspective of the customers consuming the water recovered from the ASR well, only the bulk concentration is relevant. Thus, detailed monitoring would only seem to be required if there was an indication that contaminants produced within discrete intervals would not be diluted in the bulk recovered water. Demonstrating that dilution will be sufficient to reduce concentrations may be complicated by the fact that rates of contaminant production and the extent of dilution may vary as a function of pumping rates. Furthermore, effects of heterogeneity also need to be considered for neighboring wells. If a neighboring municipal or private well is open to a more limited section of the aquifer, but one that includes a problem interval, there would be increased likelihood that consumers of water from these other wells could be exposed to contaminants at concentrations that exceed SDWA or groundwater law standards. Therefore, even if pilot studies do not indicate that water recovered from the ASR well will violate water quality standards, utilities may still need to design monitoring systems to identify the presence of problem intervals in which SDWA or groundwater law standards might be exceeded in other wells as a result of ASR operations.

Monitoring Systems and Schedules to Demonstrate Compliance with Regulatory Standards

Monitoring to comply with Underground Injection Control Program requirements and with SDWA standards for recovered water can be accomplished in a fairly straightforward manner by collecting samples at the ASR well itself prior to injection and as the water is recovered. Specific actions that the DNR should consider as mechanisms to demonstrate compliance include the following:

- Require that SDWA standards be met by the water being injected into the aquifer. In general, the SDWA prescribed monitoring schedule should be sufficient for the injected water, unless the DNR feels there is source water vulnerability to a particular contaminant or set of contaminants. A complete sanitary survey should be conducted to assess this potential for vulnerability. Also, if there is reason to believe that a particular substance may accumulate in the aquifer and increase in concentration above SDWA standards in the aquifer, then the DNR may require an alternative, lower standard for that substance during injection and an increased frequency in the monitoring of that substance. Utilities should meet all microbial filtration and disinfection standards set by the SDWA before injecting surface water into the aquifer.

- Require that SDWA standards be met by the recovered water at the entry point to the distribution system. Pilot testing may show that monitoring of some substances should be more frequent than prescribed by the SDWA standards, particularly for those substances that show significant changes in concentration during a single recovery cycle and over several recovery cycles. Similar considerations should be given to those chemicals that appear to be produced by geochemical activity in the aquifer. Additional turbidity and/or particle count standards are also recommended for a recovery-to-waste period during the initial part of the recovery cycle.

Monitoring to demonstrate compliance with the Wisconsin Groundwater Law may be more problematic since this law requires that the enforcement standards be met at any point beyond the utility's property boundary. The fact that ASR operations will cause frequent changes in directions of groundwater flow in the vicinity of the ASR well means that there may not be a single "downgradient" direction along which to install a monitoring well between the ASR well and the property boundary. Thus, strict demonstration of compliance could require numerous monitoring wells located along all boundaries of the property. Even in the case of a strong regional gradient that may constrain flow directions to a more limited range, a single monitoring well may be insufficient. The proposed locations of monitoring wells will need to be evaluated on a site-by-site basis and supported by adequate information on 1) hydraulic gradients during all phases of ASR operations and 2) potential contaminants that could be introduced to the aquifer or mobilized by ASR.

As noted in an earlier section of this report, in cases where ASR wells are located very close to a property boundary, exceedances of enforcement standards for disinfection by-products may be expected since their enforcement standards are much lower than the concentrations that would be in compliance with the Underground Injection Control Program requirements. The Wisconsin Legislature may want to consider designating alternative points of compliance, for example at distances from the ASR well equal to the expected extent of the ASR bubble. The Wisconsin Legislature may also want to consider a waiver of the enforcement standards for disinfection by-products in the case of ASR, based on consideration of relative risks of introduction of microbial pathogens to the aquifer and those associated with the disinfection by-products.

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